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Soil changes and long-term ecosystem recovery from physical and chemical load – stump harvesting and sprinkling infiltration as case studies

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Academic dissertation

To be presented, with the permission of the Faculty of Agriculture and Forestry of the University of Helsinki, for public examination in Auditorium B2, Forest Sciences Building (Viikki Campus, Latokartanonkaari 7, Helsinki) on November 9th 2018, at noon.

Title of dissertation: Soil changes and long-term ecosystem recovery from physical and chemical load – stump harvesting and sprinkling infiltration as case studies

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Dissertationes Forestales 260

https://doi.org/10.14214/df.260

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ISSN 1795-7389 (online)
ISBN 978-951-651-610-6 (pdf)

ISSN 2323-9220 (print)
ISBN 978-951-651-611-3 (paperback)

Cover photo: Lilli Kaarakka

Printers:
Unigrafia, Helsinki 2018

Publishers:
Finnish Society of Forest Science
Faculty of Agriculture and Forestry of the University of Helsinki
School of Forest Sciences of the University of Eastern Finland

Editorial Office:
Finnish Society of Forest Science
Viikinkaari 6, FI-00790 Helsinki, Finland
http://www.dissertationesforestales.fi

Kaarakka, L. (2018). Maaperämuutosten kesto ja ekosysteemin pitkän ajan toipuminen fysikaalisesta ja kemiallisesta kuormituksesta – esimerkkeinä kantojen korjuu ja sadetusimeytys. Disserationes Forestales 260. 62 p. https://doi.org/10.14214/df.260

TIIVISTELMÄ

Maaperä on metsäekosysteemin rakenteen ja toiminnan perusta. Maaperän biologiset, kemialliset ja fysikaaliset prosessit säätelevät koko metsän hiilen, ravinteiden ja veden kiertoa. Ihmisen toimenpiteet muuttavat suomalaisten metsämaiden rakennetta ja toimintaa ja edelleen metsäekosysteemin häiriödynamiikkaa.

Tässä väitöskirjassa määritettiin kahden maaperän rakenteeseen ja kemialliseen koostumukseen vaikuttavan toimenpiteen − kantojen korjuun ja tekopohjaveden muodostamisen – pitkäaikaisvaikutuksia metsämaaperän ja -kasvillisuuden rakenteeseen, toimintaan ja toipumiseen. Kantojen korjuussa maaperän pintakerros häiriintyy ja siitä poistuu hiiltä ja ravinteita korjattavien juurten ja kantojen mukana. Tekopohjaveden muodostaminen sadettamalla ravinnerikasta järvivettä harjualueille puolestaan lisää maaperään hiiltä ja ravinteita, ja käsittelyn tuloksena maaperän kemiallinen koostumus muuttuu ja metsäekosysteemi rehevöityy. Aiemmat kotimaiset tutkimukset näiden toimenpiteiden vaikutuksista on tehty korkeintaan muutamia vuosia toimenpiteiden päättymisen jälkeen eikä vaikutusten kestoa ja ekosysteemien toipumisnopeutta tunneta.

Kantojen korjuun vaikutuksia tutkittiin Keski- ja Etelä-Suomessa sijaitsevilla metsäalueilla ja tekopohjaveden ekosysteemivaikutuksia Keski-Suomessa sadetusimeytystä käyttävällä tekopohjavesilaitoksella. Tutkimuksissa tarkasteltiin metsäekosysteemin toipumista toimenpiteistä, jotka olivat päättyneet yli 10 vuotta aiemmin. Tulokset osoittavat, että kantojen korjuun ja sen jälkeen tehtävän maanmuokkauksen seurauksena kuusikoiden maaperän pintakerros häiriintyy laajalti rakenteeltaan ainakin yli 10 vuoden ajaksi. Orgaanisen aineen jakauman muuttuminen heijastuu maaperän hiilen ja typen dynamiikkaan. Suomalaisten havupuiden kannot ja paksujuuret ovat huomattava hiilen ja ravinteiden pitkäaikaisvarasto maaperässä, ja kantojen korjuun merkittävin ekologinen vaikutus on lahopuun ja sen hiilivaraston määrän väheneminen.

Tekopohjaveden muodostaminen sadettamalla muutti maaperän happamuuden ja ravinteisuuden sekä edelleen kasvilajiston, ja muutokset kestivät pitkään imeytyksen lopettamisen jälkeen. Tekopohjavesialueilla maaperän pH ja ravinteiden pitoisuudet olivat 12−15 vuoden jälkeen sadetuksen päättymisestä edelleen huomattavasti korkeampia sadetetuilla koealoilla kuin vertailualoilla. Myös aluskasvillisuuden lajien runsaussuhteet ja dynamiikka olivat edelleen muuttuneita.

Tulokset osoittavat, että maan pintakerroksen rakenne ja toiminta häiriintyy pitkäaikaisesti molempien käsittelyjen seurauksena. Ympäristövaikutusten keston ja niistä toipumisen tunteminen on ensiarvoisen tärkeää, jotta toimenpiteitä, kuten

metsäenergian korjuuta ja sen päätöksentekoa ja vastaavasti tekopohjavesilaitosten toimintaa voidaan suunnitella mahdollisimman pienin ympäristöhaitoin.

Asiasanat: metsämaaperä, maanmuokkaus, maaperän hiili, kantojen korjuu, sadetusimeytys, ekosysteemin palautuminen

Kaarakka L. (2018). Soil changes and long-term ecosystem recovery from physical and chemical load – stump harvesting and sprinkling infiltration as case studies. Disserationes Forestales 260. 62 p. https://doi.org/10.14214/df.260

ABSTRACT

Human-induced disturbances may change vegetation and carbon (C) and nitrogen (N) processes in the forest floor and the soil beneath it. The aim of this dissertation was to study the effects of physical and chemical disturbance on boreal forest soil and vegetation. The specific aims were to evaluate the rate and direction of the forest ecosystem recovery from the disturbance and to assess how C and N processes are affected by different disturbances regimes. Two contrasting soil-affecting treatments – stump harvesting and sprinkling infiltration – were studied as case studies representing a disturbance. Sprinkling infiltration alters the chemical composition of forest soil, whereas stump harvesting results in changes especially in the physical structure of the forest soil. Furthermore, in contrast to stump harvesting where C and nutrients are removed from the soil with the removed biomass, sprinkling infiltration adds large quantities of C and nutrient-rich surface water into the forest soil. As stump harvesting and sprinkling infiltration are relatively newly introduced land use practices, very little is known of their long-term effects on boreal forest soil and vegetation.

The effects of stump harvesting on forest soil surface disturbance, C and N pools and mineralization rates, understory vegetation, seedling growth and coarse woody debris (CWD) were studied in Norway spruce (*Picea abies* (L.) Karst.) stands located in Central and Southern Finland. The results of this study indicate that stump harvesting causes soil mixing and relocation of organic matter in the soil profile, which in turn is reflected to the soil C and N dynamics as soil C and N pools tended to be lower following stump harvesting. Stump harvesting combined with site preparation tends to cause more extensive soil surface disturbance than site preparation alone, and the mixing effect of stump harvesting persists on soil surface after a decade since harvest. Furthermore, this study underlines that stumps, coarse roots and fine coarse roots represent a significant portion of the CWD, belowground biomass and nutrients in a forested stand, and thus their extraction results in substantial and direct removal of biomass, C and nutrients from the stand.

The effects of sprinkling infiltration on forest soil, tree growth and understory vegetation and their respective recovery were studied in an experimental stand that had been infiltrated with surface water in order to produce artificial groundwater. The study revealed that the previously observed changes soil chemistry had persisted in the experimental stand; soil pH and base cation concentration as well as the rate of net N mineralization were still significantly higher at the infiltrated plots after a 12–15-year recovery period. These results lead to the conclusion that sprinkling infiltration results in the long-term neutralization of the forest soil. In contrast to tree growth, the

understory vegetation had not benefited from the added nutrients and organic matter, instead the large amounts of added water had created conditions unfavorable to certain plant species. In conclusion, sprinkling infiltration is an environment altering treatment which, based on the findings of this study, can have short-term effects on tree growth and long-term effects on soil processes and understory vegetation and ultimately, ecosystem recovery.

The results of this study demonstrate that disturbances affect the function and structure of forest soil and these changes can persist for at least a decade on the surface of the soil in the organic layer and deeper in the mineral soil. Furthermore, this dissertation highlights the need for long-term perspectives in ecosystem management and planning.

Keywords: artificial groundwater recharge, carbon, disturbance, ecosystem recovery, forest soil, nitrogen, stump harvesting

ACKNOWLEDGEMENTS

Me – like so many other doctoral students – would not have survived the tumultuous scientific and professional journey called "*PhD*", if it was not for the enduring and encouraging advice from our mentors. In my case this endless stream of support came from my supervisors Professor Heljä-Sisko Helmisaari and Docent Marjo Palviainen. In addition, I would like to thank Docent Aino Smolander, who guided me in many stages and phases of this adventure. Thank you all for adjusting so well to me living 9 005 km away.

This project has had many hurdles as well as successes from which the latter I owe to very many humans. I wish to extend my thanks to all my co-authors Tiina Nieminen, Leila Korpela, Antti-Jussi Lindroos, Pekka Nöjd, Mikael Marjanen, Erno Launonen, Riitta Hyvönen-Olsson, Bengt Olsson, Monika Strömgren, Tryggve Persson, Sofie Hellsten, Mikko Kukkola, Anna Saarsalmi and Janne Vaittinen. I am extremely grateful to Marjut Wallner who helped me enormously with the laboratory work. I wish to thank Department of Forest Sciences and LUKE for providing laboratory facilities for the analytical work as well as everyone at Hyytiälä Forest Station and all the individuals who helped with the field work.

Colleagues at the Department of Forest Sciences – in the decade I have spent on-and-off at the department – I have always felt appreciated and welcomed – thank you for that! Professor Pasi Puttonen – thank you for signing and stamping so many forms. My deepest appreciation is also due to my fellow PhD-students Maija Lampela, Mari Könönen, Minna Blomqvist, Maiju Kosunen and Laura Matkala – with some of you I have shared an office too. Apologies for the bad jokes and thank you for the companionship.

The immensely valuable feedback from pre-examiners Docent Jari Haimi and Professor Nicholas Clarke truly improved this dissertation. I also want to express my gratitude to Professor Dan Binkley for accepting the invitation to serve as my opponent. Professor Cindy Prescott and Professor Andrew Burton, thank you for the words of encouragement. Professor Mike Wingfield and Alexander Buck from IUFRO, thank you for providing students like me with a platform in the international forestry fora. Everyone at IFSA – keep doing what you do!

I am grateful to Maj and Tor Nessling Foundation, Niemi-säätiö, Finnish Society of Forest Science and Maa- ja vesitekniikan tuki ry. for funding this project. I also wish to extend my thanks to the AGFOREE–doctoral program, Department of Forest Sciences and Finnish Society of Forest Science for funding many of my conference trips.

Finally, I wish to thank all my friends and family for their unconditional support and understanding that life takes us places, sometimes far from home. Finally – my family, my heart and joy – Mr. Hannu and Mr. Felix – without you my days would be dull and weary.

Lilli Kaarakka, 29th August 2018, Los Angeles, California

LIST OF ORIGINAL RESEARCH ARTICLES

This dissertation is based on the following four articles, which are referred to in the text by their Roman numerals. Papers I, II and IV are reproduced with the kind permission of the publisher. Paper III is the author's version of the submitted manuscript.

I **Kaarakka, L**., Hyvönen, R., Strömgren, M., Palviainen, M., Persson, T., Olsson, B.A., Launonen, E., Vegerfors-Persson, B. and Helmisaari, H-S. (2016). Carbon and nitrogen pools and mineralization rates in boreal forest soil after stump harvesting. Forest Ecology and Management 377: 61–70 https://doi.org/10.1016/j.foreco.2016.06.042

II Hyvönen, R., **Kaarakka, L.**, Leppälammi-Kujansuu, J., Olsson, B.A., Vegerfors-Persson, B., Palviainen, M. and Helmisaari, H-S. (2016). Effects of stump harvesting on soil C and N stocks and vegetation 8–13 years after clear-cutting. Forest Ecology and Management 371: 23–32. https://doi.org/10.1016/j.foreco.2016.02.002

III **Kaarakka, L.,** Smolander, A., Lindroos, A-J., Nöjd, P., Korpela, L., Nieminen, T.M. and Helmisaari, H-S. Sprinkling infiltration as an artificial groundwater recharge method – long-term effects on boreal forest soil, tree growth and understory vegetation. (Submitted for review at Forest Ecology and Management)

IV **Kaarakka, L.**, Vaittinen, J., Marjanen, M., Hellsten, S., Kukkola, M., Saarsalmi, A., Palviainen, M. and Helmisaari, H-S. **(**2018). Stump harvesting in *Picea abies* stands: Soil surface disturbance and biomass distribution of the harvested stumps and roots. Forest Ecology and Management 425: 27–43. https://doi.org/10.1016/j.foreco.2018.05.032

Lilli Kaarakka is fully responsible for the summary of this dissertation. In papers **I**, **III** and **IV,** Lilli Kaarakka is the corresponding author and responsible for the writing and interpretation of the results. Study **IV** is based on data partly collected by others. In the studies presented in papers **I**, **II** and **III**, Lilli Kaarakka planned the experimental work together with the co-authors and participated in all the field work and completed the laboratory analyses. In papers **I** and **II**, parts of the statistical analyses were completed by the co-authors at SLU.

LIST OF ABBREVIATIONS

C	carbon
CWD	coarse woody debris
CEC	cation exchange capacity
DOC	dissolved organic carbon
MRT	mean residence time
N	nitrogen
R_h	heterotrophic respiration
OM	organic matter
S	slash harvesting (= logging residue harvesting)
SOC	soil organic carbon
SOM	soil organic matter
SOH	stem-only harvesting
SS	slash and stump harvesting
WTH	whole-tree harvesting (stem + logging residue)
WTH + S	whole-tree harvesting combined with stump harvesting

Interchangeable terms

The completion of this dissertation and its findings are a result of co-operation between different research institutions, between which some of the terminology used in forest research may differ. Thus, to avoid discrepancies and confusion about terminology, it seems fitting to define a few interchangeable terms. In this dissertation, soil organic layer is discussed as a whole and I do not distinguish between the organic layer sub-layers. In the published articles included in the dissertation the organic layer is referred to as the *humus layer*. Furthermore, the term *slash* refers to logging residues and is used in one the articles (study **II**). In studies **I** and **II**, the stand-level units are referred to as "*experimental stands*" and "*clear-cut stands*", respectively, and the experimental units within the stand are "*experimental subplots*". In study **IV,** the stands are referred to as "*experimental sites*" and the experimental units within the site are "*experimental plots*". Finally, within the context of this dissertation, terms "*field layer vegetation*", "*ground vegetation*" and "*understory vegetation*" are used interchangeably.

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1 INTRODUCTION

1.1 Boreal forest management and soil carbon

Boreal forest is the largest land biome in the world (Astrup et al., 2018). Two-thirds of the boreal forest area is being managed for purposes such as timber production, fire protection and conservation (Gauthier et al., 2015; Astrup et al., 2018). Due to climate change, boreal forests are faced with unprecedented changes, which can threaten the resilience of ecosystems and potentially adversely affect the ecosystem services provided by the area (Gauthier et al., 2015).

Forests cover 86 % of the land surface in Finland out of which 78 % (20 million hectares) are considered to be productive forests and are intensively managed (Kaila and Ihalainen, 2014). Finland, like other Fennoscandian countries, has a long tradition of utilizing forest-based biomass for energy and industry purposes, and the use has increased in the last decades due to changes in international and regional energy policies (Ericsson et al., 2004; Helmisaari et al., 2014). More intense logging operations, in which increasingly more forest biomass is harvested from the stand are becoming common in the region. Thus, whole-tree harvest (WTH), in which commercial stem and crown biomass and eventually also stumps are removed from site, is becoming an alternative to conventional stem-only harvest (SOH). Forest biomass used for bioenergy purposes comes directly from the forest, unprocessed (Helmisaari et al., 2014). The use of wood chips (which consist of logging residues: branches, tree tops, stumps, small-diameter trees and defect stemwood) has more than doubled between the years 2006–2016, from 3 million m^3 to 7.4 million m^3, accounting for 37 % of the wood-based fuel use in 2016 (Kortesmaa et al., 2017; LUKE, 2017). The Finnish government has planned to increase the use of forest chips to 13.5 million m^3 by the year 2020 to meet the EU 2020–energy policy targets (Laitila et al., 2008a; LUKE, 2017). In practice, this would mean that logging residues should be collected from most clear-cuts, as 15.3 million m^3 of logging residues (excluding stumps) have been considered to be technically harvestable (Asikainen et al., 2008).

The climate benefits of intensive forest harvesting have been questioned, however. Intensive biomass harvesting from a forest stand results in both direct – combustion – and indirect carbon (C) emissions – reduced C stock of the decomposing biomass (Repo et al., 2011; Zanchi et al., 2012). In other words, C allocated to woody biomass will be released immediately to the atmosphere instead of being retained in the ecosystem for a long time. Consequently, the choice of the compartment of forest biomass used for bioenergy purposes greatly affects the magnitude and timing of the potential C losses associated with harvesting (Repo et al., 2011; Repo et al., 2012; Repo et al., 2015).

In an undisturbed forest ecosystem, C and nutrients are effectively recycled and only small amounts is lost through leaching and run-off, erosion and volatilization.

Disturbances, be it human-induced or natural, chemical or physical, shape the composition and function of the forest stand. Varying greatly in intensity, scale and frequency, disturbances create heterogeneity in the landscape (Kuuluvainen et al., 2012; Seidl et al., 2017) and can result in losses of both vegetation C and soil C (Deluca and Boisvenue, 2012). The cycle of disturbance and recovery, however is an essential part of the boreal forest stand (e.g. Binkley and Fisher, 2012).

Forests play an important role in converting C from the atmosphere into ecosystem C which is deposited in vegetation and soil. Soil in turn provides the foundation for plant and tree growth in a forested ecosystem. Boreal forest soil represents one the largest pools of C in the northern hemisphere (Mahli et al., 1999; Hyvönen et al., 2007; Deluca and Boisvenue, 2012; Amundson et al., 2015; Astrup et al., 2018). Reports estimate that as much as 60–70 % of the ecosystem C is stored in the soil in boreal forest stands (Pan et al., 2011). Compared to the temperate and tropical forests, physiological processes in the boreal forests are limited by the cold climate and the short growing season. Species adapted to strong seasonal changes and at times harsh climate, such as conifers, thrive in the region. Many of the soil biological processes occur in the surface layers of the soil, i.e. the topsoil, but subsoil holds an important role in biogeochemical cycling and importantly, in groundwater dynamics. Boreal upland forest soils are characterized by a visible organic layer, often called the O-horizon, which covers the mineral soil, and is further divided into sub-layers that differ in the degree of decomposition (Brady and Weil, 2008; Binkley and Fisher, 2012). Soil organic matter (SOM) originates from both aboveground and belowground parts of forest organisms, and is made up of organic materials in various stages of decomposition with an average C content of ~ 47–50 % (Brady and Weil, 2008). Thus, SOM is a key component in the C cycle of a forested ecosystem and a driver of numerous processes in the soil and essential for long-term site productivity (e.g. Prescott et al., 2000; Schmidt et al., 2011; Binkley and Fisher, 2012; Lehmann and Kleber, 2015).

Both abiotic and biotic processes contribute to the formation and evolution of SOM in the soil. In a coniferous forest stand, the organic layer represents a significant pool of SOM, but recent research has also highlighted the importance of mineral soil as a long-term storage of C (Finér et al., 2003; Rumpel and Kögel-Knabner, 2011; Merilä et al., 2014). Organic matter (OM) is added to the soil surface with litter and through leaching and physical processes is incorporated deeper into the soil. Root and fungal litter and exudate inputs, as well as soil forming processes and soil texture, have an impact on the composition of OM in the deeper mineral soil layers (Rumpel et al., 2002). Further, in boreal forest soils, the turnover time for soil organic carbon (SOC) varies vertically from years to decades in the organic layer, to millennia deeper in the mineral soil (Trumbore, 2000; Fröberg et al., 2011; Schmidt et al., 2011). A study based on data from Finland, Sweden, Denmark and Norway by Callesen et al. (2003) reported that SOC pool was correlated with climatic factors (temperature and precipitation) on

coarse- to medium-textured soils but on finer textured soils, SOC varied independently of these two. This indicates that soil texture too is an important determinant of SOC stocks. Finally, the source of SOM determines the decomposition rate; variation in decomposability of leaf litter, roots and coarse woody debris (CWD) is a function of the physical and chemical composition of the source (Hansson et al., 2013). CWD, consisting of dead tree parts from both above- and belowground in various stages of decomposition, is an important component in the forest C and N cycles (Rabinowitch-Jokinen and Vanha-Majamaa, 2010; Magnusson et al., 2016).

1.2 Soil nutrient dynamics

Although boreal soils contain large amounts of nitrogen (N), only a small fraction of this pool is in readily mineralizable form and directly available to plants (Korhonen et al., 2013). In an undisturbed forest stand, most of the available N is tightly recycled within the plant-soil-microbe sphere (Paavolainen, 1999). All the N transformations in the soil; depolymerisation of high-molecular mass organic N, mineralization, immobilization, nitrification and denitrification, are driven by microbial processes. Plant-available N is added to the soil slowly through the depolymerization of large N-containing compounds in the decomposing litter and organic matter to small-molecular-size N compounds and finally by N mineralization which is controlled by pH, moisture conditions, temperature and litter quality. In current research, depolymerization, enabled by microbial enzymes is viewed as the bottleneck in plant N availability (Schimel and Bennett, 2004; Wild et al., 2015; Högberg et al., 2017). Nonetheless, as highlighted by research focusing on boreal tree species the relationship between trees and the soil microbes is highly complex; in part beneficial, competitive, or both, and these dynamics are likely to vary between different tree species and sites (Priha and Smolander, 1999; Smolander and Kitunen, 2011). In Finnish forest soils, the microbial biomass N pool varies 3–9 % of total soil N depending on the site fertility, tree species and development stage of the stand (Priha and Smolander, 1997; Priha and Smolander, 1999; Smolander and Kitunen, 2011). Recent studies have reported that plant-induced compounds (enzymes, terpenes) can potentially induce the decomposition in the soil N recalcitrant pool (Kieloaho et al., 2016) or inhibit the N mineralization processes in the soil (Smolander et al., 2012) thus potentially influencing the soil N pool.

The processes in which N is traded between plants and soil microorganisms, such as the symbiotic ectomycorrhizal fungi or microbes, are driven by the competition for N, particularly in N-limited ecosystems, such as the boreal soil (Sponseller et al., 2016). When the soil available pool of N is low, biotic retention of N by soil microorganisms, who themselves can be N-limited, can further intensify the decline of plant-available N in the soil. Increased immobilization of N can trigger plants to allocate more C belowground, further exacerbating the microbial N sink, thus creating a positive feedback loop (Näsholm et al., 2013; Högberg et al., 2014; Sponseller et al., 2016;

Högberg et al., 2017), but if and how this effect is translated into tree growth, remains to be observed. Finally, it is possible that in high-latitude ecosystems, microbes in the deeper mineral soil horizons are more C-limited than in the organic layer, where the low availability of N is limiting microbial N transformations (Wild et al., 2015).

Nitrification, i.e. the production of nitrite and nitrate, is limited by low pH in boreal coniferous forest soils and is typically negligible unless N is added to the system (fertilization/atmospheric deposition) (Paavolainen and Smolander, 1998; Sponseller et al., 2016). Increased nitrification and the consequent leaching of nitrate are associated with soil acidification, particularly in N-saturated soils (Högberg et al., 2017). Atmospheric deposition of N (wet+dry deposition) in Finland varies between 1–7.5 kg ha^{-1} yr^{-1} in a north-south gradient (Korhonen et al., 2013; Dirnböck et al., 2014; Palviainen et al., 2017), averaging 2.8 kg ha^{-1} yr^{-1}, which compared to central European countries is low (Mustajärvi et al., 2008; Dirnböck et al., 2014). Nevertheless, atmospheric deposition represent a major source of N to a forest stand, particularly in northern latitude forests (Korhonen et al., 2013; Palviainen et al., 2017).

In acidic boreal soils, the abundance of cations necessary for plant growth, such as calcium (Ca) and magnesium (Mg), is very much linked to the OM quantity and quality of the soil (Ross et al., 2008). Due to its colloidal structure OM has a high cation retention capacity, in other words, the capacity to hold and exchange cations and anions from the soil solution. Cation exchange capacity (CEC) of OM is very much pH-dependent, but when cations are present in equivalent amounts in the soil solution (in neutral pH), the order of strength of affinity in the exchange sites is $Al^{3+} > H^{+} > Ca^{2+} > Mg^{2+} > K^{+} = NH_4+ > Na^{+}$, owing to their relative valences (Brady and Weil, 2008). Cations, such as Ca^{2+} and K^{+} and Mg^{2+}, are more easily displaced from the negatively charged exchange sites and thus available to plants and microorganisms. Soil pH, in turn, is controlled by the hydrogen ions entering the soil through precipitation, root exudates, mineral weathering and leaching from vegetation.

1.3 Forest management and physical disturbance

Productive forest stands in Finland are intensively managed. Even-aged forest management cycle is characterized by stands with long rotation times (55–100 years depending on the tree species and site properties) and clear-cutting at the end of rotation, followed by a re-establishment of the stand through planting or natural regeneration. In addition, mechanical site preparation is carried out in most forest stands. Mounding, in which soil is inverted with an excavator to form a mound, is the most common method used in Finland for planting Norway spruce (*Picea abies* (L.) Karst.) (Kortesmaa et al., 2017). In site preparation, mineral soil is exposed to various depths and SOM originating from the surface is mixed with mineral soil material. Post-mounding soil landscape includes three surfaces; the mound with mineral soil mixed with organic matter on top, the pit where the mound material has been scooped from and finally, the

undisturbed surface. Thus, soil preparation changes the temperature and water conditions of the soil surface layers by creating elevated mounds of soil and shaded pits, which have a different microclimate and organic matter distribution than that of the undisturbed soil (Pumpanen, 2008). Studies conducted in temperate forests have given indication that C stored in the mineral soil seems to be less sensitive to physical disturbances than the C organic layer owing to the more less-stabilized and volatile nature of C in the latter (Jandl et al., 2007; Nave et al., 2010), however this has not been universally confirmed nor rejected by studies in the boreal region (Thiffault et al., 2011).

In Finland, 80 % of the managed forest stands are dominated by two coniferous species; Scots pine (*Pinus sylvestris*) and Norway spruce (*Picea abies*) (Kortesmaa et al., 2017). Trees affect the chemical and physical composition of forest soil in numerous ways (Hansson et al., 2011; Binkley and Fisher, 2012), however, the effect of specific tree species on soil processes – although extensively studied (Smolander and Kitunen, 2011) – has remained somewhat of a mystery. Nevertheless, Swedish and Finnish studies have given indication that tree species do affect soil pH, N processes, C pools and soil biota, both directly though litter production (above- and belowground) and indirectly through influencing the abiotic factors of the stand (microclimate, canopy size) (Priha and Smolander, 1997; Hansson et al., 2011; Smolander and Kitunen, 2011; Hansson et al., 2013). Annual litterfall varies according to tree species. A modeling study by Saarsalmi et al. (2007) concluded that the annual litterfall of Norway spruce is dependent on latitude, temperature and height of the trees in the stand. Compared to Scots pine and birch stands, spruce stands have higher litter production rates and thus thicker organic layers (Palviainen et al., 2004; Hansson et al., 2011) in similar climate.

1.3.1 Harvest intensity and soil disturbance

Forest management plays an important role in directly controlling the stand biomass stock (e.g. Jandl et al., 2007; Hyvönen et al., 2007) thus affecting the soil C and N pools and site productivity. Whether stump harvesting affects stand productivity is under debate. In Finland, stump harvesting started in 2000 and peaked in 2010–2013 with 1.1 million m^3 annually, the current annual harvest being 0.76 million m^3 (LUKE, 2017; Persson and Egnell, 2018). Stump harvesting is currently only practiced in fertile and moderately fertile Norway spruce (*Picea abies* (L.) Karst.) stands (site types OMT–MT, (Cajander, 1949)).

The effects of harvesting on forest soil are very much dependent on the location, developmental stage (age) and structure of the stand, as well as the intensity of the harvesting operation, and the method chosen for site preparation. Nevertheless, it should be stated that forest harvesting (combined with site preparation) *per se* causes a disturbance to the forest floor and exposure of the mineral soil (Finér et al., 2003; Kataja-aho et al., 2011a; Tarvainen et al., 2015), however the intensity and scale of the disturbance determines how it affects decomposition and mineralization processes in

the soil (Smolander et al., 2000; Yanai et al., 2003a; Jandl et al., 2007; Hope, 2007; Kreutzweiser et al., 2008; Nave et al., 2010; Harmon et al., 2011; Helmisaari et al., 2014; Clarke et al., 2015; Piirainen et al., 2015). Site preparation is practically always carried out following stump harvesting to create more favorable planting spots for conifer seedlings (Saarinen, 2006; Laitila et al., 2008b; Rantala et al., 2010; Saksa, 2013). Past studies have given indication that stump harvesting causes physical disturbance to soil (Walmsley and Godbold, 2010). In Finland and Sweden, stump harvesting combined with site preparation has been reported to result in 70–80 % of the soil surface area being disturbed, in comparison with 25–50 % when only site preparation is carried out (Kataja-aho et al., 2012b; Strömgren and Mjöfors, 2012; Tarvainen et al., 2015). Stump harvesting causes heavy traffic at the logging site, as logging equipment is hauled to and from the stand, thus potentially resulting in more soil disturbance. Berg et al. (2015) reported surface disturbance of 59–61 % post-stump harvesting combined with mounding, but concluded "that much of the ground disturbance is associated with the creation of wheel ruts rather than stump harvest per se." Tarvainen et al. (2015) also reported an increase in soil surface disturbance and exposed mineral soil following stump harvest, but they too acknowledged that some study sites were more heavily disturbed by logging machinery.

Site preparation and the consequent soil mixing can affect the soil seed bank. Post-mounding micro landscape can be beneficial to some plants; the exposed mineral soil patches act as refuges and germination beds for plant seeds (Kataja-aho et al., 2011a). Stump harvesting has been reported to improve planted seedling survival and natural regeneration, particularly that of birch (*Betula* sp.) (Karlsson and Tamminen, 2013; Saksa, 2013; Tarvainen et al., 2015), possibly owing to the larger areas of exposed mineral soil (Kataja-aho et al., 2011a). Kataja-aho et al. (2012a) observed a small (10 %) growth increase in Norway spruce seedlings three years after stump harvesting but concluded that this effect could be transient. Several studies have described the rapid growth of early successional plant species (grasses such as *Deschampsia flexuosa*) as a response to harvesting (Olsson and Staaf, 1995; Bergstedt and Milberg, 2001; Palviainen et al., 2005a; Bergholm et al., 2015; Tonteri et al., 2016), thus the competition for resources (particularly for N) can affect tree seedling growth. Page-Dumroese et al. (1998) reported a 20 % reduction in seedling height and 30 % reduction in seedling root collar diameter on stump harvested sites for three year-old Douglas-fir (*Pseudotsuga menziesii* var. *glauca*) in the US. Nevertheless, there is a lack of long-term studies addressing how stump harvesting affects forest soil and tree growth in the boreal region, as the observed time period in current studies ranges from 15 to 30 years (Egnell, 2017; Persson et al., 2017). Long-term studies on the effects of stump harvesting on tree growth are particularly important considering that effects of intensive harvesting might be lagged and potentially only observed at the end of the stand rotation (Thiffault et al., 2011; Egnell, 2017).

1.3.2 Biomass distribution of harvested stump-root systems

Coarse roots, which together with the stumps form the stump-root system, are a significant belowground biomass component of the tree and contribute to soil physical stability. In fact, defining where the stump ends and coarse roots begin, can be challenging as the dimensions of stump-root system are complex and highly variable (Kalliokoski et al., 2008). Stump and large diameter coarse roots are the largest CWD component in a managed boreal forest, as other types of CWD are extracted in forestry operations (Eräjää et al., 2010; Rabinowitch-Jokinen and Vanha-Majamaa, 2010; Walmsley and Godbold, 2010; Palviainen et al., 2010). The volume of low stump CWD on clear-cuts and young managed forests can be 2.5–4 times greater than that of other types of CWD (logs, branches, snags) (Eräjää et al., 2010; Anderson et al., 2015).

Due to their relatively slow decomposition process, stumps and coarse roots serve as long-term C and N pools and as sources of nutrients in a forest stand (Sucre and Fox, 2009; Melin et al., 2009; Palviainen et al., 2010; Hellsten et al., 2013; Palviainen and Finér, 2015). Considering that the stand rotation times for conifers in Finland and Sweden are typically more than 65 years, this slow-release C and N could be imperative for site productivity. Only a handful of studies have attempted to estimate the biomass and N removals associated with stump and coarse root removal (e.g. Hakkila, 1975; Brassard et al., 2011; Augusto et al., 2015; Palviainen and Finér, 2015) due to the arduous nature of sampling entire stumps-root systems. In Finland, a study compiled from data from over 400 conifer stump-root systems sampled across the country estimated that stumps and coarse roots (diameter > 5 cm) comprised 26−34 % and 68 % of the entire stump-root biomass in a mature Norway spruce stand, respectively (Hakkila, 1975).

1.4 Sprinkling infiltration and chemical disturbance

Water is a resource that is deficient in many parts of the world. In Fennoscandia, there is plenty of surface water available for household use but to make the water potable it's treated in a variety of ways to remove organic C. Groundwater, naturally filtered through soil layers has a low organic C content but reserves are often scattered and not large enough to accommodate larger cities. Thus, many cities in Finland are using or planning to produce artificial groundwater by infiltration (Kätkö et al., 2006). Sprinkling infiltration differs from other infiltration methods in that surface water (i.e. lake water) is sprinkled directly onto the forest floor via a network of pipes and in contrast to basin recharge, it does not require an extensive land area to be cleared (Helmisaari et al., 2003). During sprinkling infiltration, forest soil is subjected to extremely large inputs of water, in which relatively large quantities of nutrients such as Ca^{2+}, Mg^{2+} and N are added onto the forest floor and into the soil, chemically changing it. Thus contrary to harvesting, sprinkling infiltration causes a chemical rather than a

physical disturbance in the soil. Artificial groundwater recharge in the form of sprinkling infiltration has been introduced in Finland relatively recently. This current experiment described in this dissertation serves as the only long-term study in the region on the topic.

Only a very few studies in Fennoscandia have studied the effects of added water on forest soil and tree growth (Paavolainen et al., 2000a; Paavolainen et al., 2000b; Lindroos et al., 2001; Nöjd et al., 2009). These previous studies in Finland have demonstrated a relatively instant ecosystem response to sprinkling infiltration and reported changes in soil pH, N transformation processes, base cation pools and tree growth following the treatment (Paavolainen et al., 2000a; Paavolainen et al., 2000b; Lindroos et al., 2001; Nöjd et al., 2009). These studies have thus given indication that sprinkling infiltration leads to the neutralization of the forest soil (Paavolainen et al., 2000a; Paavolainen et al., 2000b; Lindroos et al., 2001). This shift in soil acidity, together with the added N, in turn acts as a driver for acid-sensitive N transformations such as nitrification (Paavolainen et al., 2000a).

Little is known on the effects of sprinkling infiltration – or irrigation in general – on forest understory vegetation even though it is very likely that adding large quantities of surface water and nutrients to the forest floor affects the vegetation in the stand. Vegetation responses to environmental change depend on the nature and intensity of the disturbance – be it clear-cutting or irrigation – and on the environmental requirements of the species in question.

Understory species competition possibilities, based on their specific requirements, change after any disturbance which alters the availability of resources such as light, water or nutrients. Even if the impacts of different disturbances vary depending on the resources they affect, their comparison may reveal also similarities. The effects of forest management operations, such as thinnings and clear-cuts, have been extensively studied in the region however, and forest management has been considered to be the principal driver of changes in the relative abundance and cover of boreal plant species (Palviainen et al., 2005b; Tarvainen et al., 2015; Tonteri et al., 2016). Furthermore, harvesting intensity determines the main change in ground vegetation dynamics (Bergholm et al., 2015; Tonteri et al., 2016). Typically the abundance of early successional light-demanding species increases and late successional shade-tolerant species decreases after clear-cutting (Bergstedt and Milberg, 2001; Tarvainen et al., 2015; Tonteri et al., 2016). Plants that have been shown to increase with harvesting intensity include forbs and grasses such as *Epilobium angustifolium* and *Avenella flexuosa (*also known as *Deschampsia flexuosa),* whereas dwarf shrubs (*Vaccinium* spp.), and mosses and lichens have been reported to decrease in abundance following clear-cutting (Palviainen et al., 2005b; Hedwall et al., 2013; Tonteri et al., 2016). A study by Strengbom et al. (2001) suggests that the time needed for recovery of the ecosystem biota may be substantial in originally N-limited ecosystems that had been fertilized. They studied previously fertilized experimental plots in Northern Sweden and

found that the recovery of the understory vegetation following a N-induced vegetation shift is a slow process and that nine years after the termination of the treatment no changes were detected in plant diversity (Strengbom et al., 2001).

Forest ecosystem recovery from disturbances depends on numerous factors; whether it is human-origin or of natural source, and on the intensity and scale of the disturbance (Niemelä, 1999). In addition, understanding the structure and function of the natural and undisturbed forest stand forms the necessary foundation to which the effects of the disturbances should be reflected (Kuuluvainen et al., 2002). Soil and vegetation recovery period depends on several factors, which can return the soil closer to its original structure and function. Post-disturbance emergence of reestablishing vegetation contributes to the re-formation of the soil layer structure and improves the C and nutrient status of the soil and thus potentially helps to restore the physical and chemical functions disturbed by the harvest. In sprinkling infiltration, recovery occurs slowly over time as hydrogen ions in precipitation acidify the surface soil layers and slowly transform the soil acidity balance to its original state. More studies have been done on ecosystem changes caused by disturbance than on assessing ecosystem recovery in space and time.

2 OBJECTIVES AND AIMS

The aim of this thesis was to assess the effects of certain human-induced physical and chemical disturbance on forest soil. The specific goal was to evaluate the rate and direction of the recovery of the ecosystem from a disturbance, and to estimate the longevity of the changes in the forest soil and vegetation.

More specifically, the objective was to study the effects of two soil treatments practiced in Finland – stump harvesting and sprinkling infiltration (i.e. groundwater recharge with lake water) – on soil C and nutrient pools, acidity, surface disturbance and soil compaction, tree growth and vegetation dynamics, and how long it takes for the soil and vegetation to recover from these disturbances. Due to their intensity, it seems probable that infiltration and stump harvesting both affect the C and nutrient processes in the soil. However, these two treatments affect forest soil differently physically and chemically. While stump harvesting causes a mechanical soil disturbance by removing biomass from soil and mixing soil top layers, the physical disturbance agent in sprinkling infiltration is the sprinkled water with a high surface load. Sprinkling infiltration adds large quantities of C and nutrients in water into the forest soil whereas stump harvesting results in the removal of biomass and nutrients. Since both of these practices were introduced relatively recently (in the last 15–20 years), from a time-perspective, this dissertation represents one of the first studies to assess the long-term effects of these two treatments.

The specific objectives of this study were:

- to investigate the effects of stump removal on soil C, N and nutrient pools, C and N transformations (studies **I** and **II**), the amount of CWD and the abundance of young trees and understory vegetation (study **II**),
- to estimate how much soil surface disturbance is caused by stump harvesting (studies **II** and **IV**), if the disturbance effect persists over time (**IV**)
- to assess/estimate how much biomass and N is removed at stump harvesting with different stump-root system compartments (study **IV**),
- to study the response of forest soil and vegetation to sprinkling infiltration and the rate and direction of the recovery of the ecosystem from it (study **III**),
- assess the legacy of the disturbances (studies **I**,**II**,**III**,**IV**)

The main hypothesis of this thesis is that 1) soil and vegetation recovery depends on the physical and chemical properties of the disturbance; represented in this study by stump harvesting and sprinkling infiltration. The changes in soil and vegetation are dependent on the treatment nature and intensity, and characteristics of the stand, which together also determine how fast the ecosystem recovers from a disturbance. Specifically, we hypothesize that 2) stump harvesting results in increased soil surface disturbance and soil mixing, which in turn can influence the soil C and N processes. In addition, we expect to confirm that substantial quantities of biomass and N are removed from the soil with the coarse roots that are pulled along with the stump. In addition, we hypothesize that 3) soil recovery at the infiltrated sites occurs relatively slowly over time as hydrogen releasing compounds in precipitation, throughfall and organic leachate from the surface of the soil acidify the soil layers and slowly transform the soil cation exchange capacity to its original state. Finally, we hypothesize that sprinkling infiltration causes changes in the understory vegetation composition.

3 MATERIALS AND METHODS

This section provides an overview of the experimental sites, sampling and laboratory analyses. More detailed description of the methods used can be found in the articles (**I–IV**) included in this dissertation.

3.1 Stump harvesting study sites

The effects of stump harvest on soil disturbance were studied at four clear-cut Norway spruce sites in Southern and Central Finland (Table 1, Figure 1). Logging residues were harvested at all the sites following clear-cutting, thus all the experimental sites had been subjected to whole-tree harvesting (WTH). Stump harvesting (study **I** = WTH+S, studies **II**, **IV** = SS) was performed in half of the experimental plots and half were subjected to mounding only (study **I** = WTH, study **II** = S, study **IV** = M). Site preparation in the form of mounding (i.e. the excavator scoops topsoil from one patch to another, thus creating a mound and a pit) was carried out in all clear-cuts, also where the stumps were removed. Stumps were harvested with an excavator equipped with a stump bucket, which splits the stump in half before lifting it.

Two of the experimental locations, Honkola and Haukilahti, were used to study the effects of stump harvesting on soil C and N pools and mineralization, the abundance of CWD, soil surface disturbance and understory vegetation and seedling growth (studies **I**, **II**). The two experimental sites in Honkola were located approximately 300 m apart, while the six sites in Haukilahti were located within a 4 x 4 km area. Soil surface disturbance measurements were additionally carried out in Hyvinkää and Karkkila, where the sites were approximately 300–600 m apart (study **IV**).

3.1.1 Soil surface disturbance, soil sampling and analyses

At the time of sampling, three 30 m x 30 m (900 m^2) experimental plots at each experimental site (known as "*stand*" in studies **I**,**II**) were established, altogether 6 in Honkola and 18 in Haukilahti (studies **I**,**II**). Soil samples were collected from the experimental plots in the summers 2012 and 2013 from the organic layer and mineral soil for analyzing soil nutrient and C contents, mineralization and respiration processes. At Honkola, soil samples were in addition collected to study the N transformations in the soil. During soil sampling, soil surface disturbance classes were estimated with a soil corer. Three disturbance classes were identified; (i) mound, (ii) undisturbed soil surface and (iii) pit (Table 2, Figure 2). In addition, the extent of soil surface disturbance was assessed at three of the stump harvested sites (Haukilahti, Karkkila and Hyvinkää) in 2014 (study **IV**). One of the sites for soil surface disturbance studies, Karkkila, was the site for biomass sampling, carried out in December 2007 (study **IV**).

Table 1 Site characteristics of the stump harvested experimental sites. Annual mean temperature and precipitation is given for 1981–2010 (Pirinen et al., 2012). Effective temperature sum (degree days, d.d.) is the sum of daily mean temperatures above +5°C for 1961–2016 (FMI, 2018). Forest site types follow the classification system by Cajander (1949).

Location	Haukilahti	Karkkila	Hyvinkää	Honkola
Coordinates	61°48'N, 24°46'E	60º35'N, 24º13'E	60°38'N, 25°01'E	61º09'N, 23º25'E
Mean annual temperature °C	3.8	4.6	4.6	4.6
Effective temperature sum (d.d. above 5 °C)	1250	1400	1350	1320
Precipitation (mm)	643	647	660	627
Soil type	Sandy loam	Silt loam	Sandy loam	Loamy coarse sand
Year of harvest	2001−2002	2007	2010	2001
Harvested stem volume (m^3 ha^{-1}) at clear-cutting	270	400	230	303–405
Forest site type	*Vaccinium myrtillus (MT)*	*Vaccinium myrtillus* (MT)	*Vaccinium myrtillus* (MT)	*Oxalis acetocella-V. myrtillus* (OMT)

Table 2 Soil surface disturbance classes and their definitions. Disturbance class was determined with a soil corer.

Disturbance class	Definition
Undisturbed	Intact humus layer. No signs of soil surface disturbance or mixing of soil layers.
Pit	Humus layer absent. Exposed mineral soil. Vertically lower than the undisturbed soil.
Mound	Exposed mineral soil due to soil inversion with an excavator. Humus layer deeper in the mound. Vertically exposed environment.

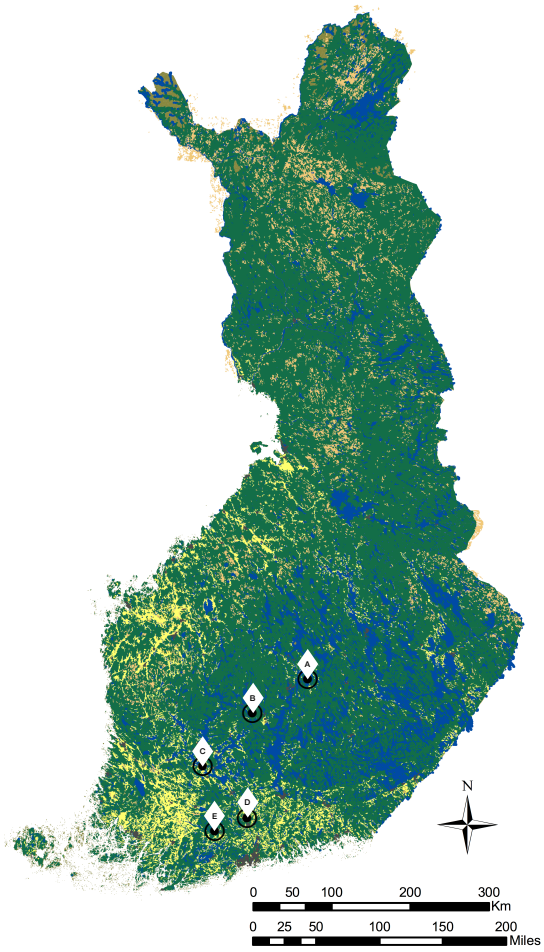

Figure 1 The experimental sites used in this study; sprinkling infiltration (A – Vuontee) was studied at one location and stump harvesting at four locations (B–Haukilahti, C– Honkola, D – Hyvinkää, E – Karkkila) in Central and Southern Finland.

The soil samples from Haukilahti and Honkola were stored at +5°C prior to chemical analyses. To homogenize the soil material, the humus samples were sieved through a 6-mm sieve and the mineral soil through a 2-mm sieve. This method also removes bigger live roots and coarse plant remnants (studies **I**,**II**). Soil C and N concentrations (study **II**) were measured directly from the air-dried samples with a VarioMax CNS-analyzer. Total pools of N and C were calculated using the formula:

$$\text{Pool (g m}^{-2}\text{)} = \text{Concentration (mg g}^{-1}\text{ soil)} \times \text{BD}_{<2}\text{ (g cm}^{-3}\text{)} \times \text{layer thickness (cm)} \times 100,$$

where $BD_{<2}$ is the measured bulk density of the 0–2 mm fraction. Pools were corrected for the field estimated soil stone content. Total pools (g m^{-2}) were only determined for the experimental plots at Haukilahti, because sample volume was not recorded at Honkola.

Figure 2 The different soil surface disturbance classes; undisturbed (top), pit (center) and mound (bottom). The soil surface layer structure in the undisturbed soil consist of an intact organic layer and mineral soil layers. The soil cores illustrate that the dark organic layer is missing entirely from the pits and in the mound, organic matter has been mixed with the mineral soil. (Photos: Heljä-Sisko Helmisaari and Mikael Marjanen)

To determine the rates of C and net N mineralization (study **I**), fresh soil samples were incubated for 26 days at a constant temperature of +15°C. CO_2-concentrations were measured every week and averaged for the 26-day-period. To calculate net N mineralization and nitrification, initial (NH_4+NO_3)- or NO_3-N concentrations were subtracted from the final (post-incubation) concentrations, respectively. To determine the mean annual heterotrophic respiration (R_h) the estimates of the laboratory measured C mineralization rates were extrapolated to the field by multiplying the C mineralization rates obtained at +15 °C and 60 % water-holding capacity for each incubated sample (i.e. each plot, disturbance class and soil layer) by the amount of C per plot (g C m^{-2}), disturbance class and soil layer and the number of days per year. Both C and N mineralization rates were normalized to the C concentration in the soil.

3.1.2 Mapping of CWD, understory vegetation and young trees

The mapping of the CWD (including stumps, logs and other types of small wood) was completed together with the surveying of the understory vegetation in 2013–2014 at Honkola and Haukilahti (study **II**). Circle plots (diameter = 6 meters) were used to estimate the amount of CWD and the number of young trees on each 30 m x 30 m experimental plot. The root collar girths of all young trees (planted and naturally regenerated) were also measured. The cover and presence of field layer species and mosses were determined with a point-intercept method using 50 cm x 50 cm quadrats.

3.1.3 Excavation of stumps and coarse roots

Stump-root system biomass measurements were carried out at Karkkila in December 2007 (study **IV**). One stump-harvested 30 m x 30 m experimental plot was used for stump biomass sampling after clear-cutting. The experimental plot had 33 trees (367 ha^{-1}) before clear-cutting. In total 26 stump and root systems, including both coarse roots (diameter > 35 mm) and thin coarse roots (diameter = 5–35 mm), were excavated. The diameter and height of each stump was measured before extraction. Each extracted stump was weighed at the field site. In addition, coarse roots and thin coarse roots were separately weighed from 17 trees. Stump sector (SS), stump discs (SD) and coarse root samples (CR) were collected from the pulled stump-root systems. All the samples included bark. A few (1–3) of the smaller coarse roots (TCR; diameter 5–35 mm) were sampled in their entirety (i.e. the whole root was collected). The samples were sealed in plastic bags and stored in a freezer until further analyses.

The volumes of the collected stump and coarse wood samples were determined gravimetrically with the water displacement method (Olesen, 1971). To determine the dry mass (kg), the samples were dried in 70 °C for 1–5 days, depending on the size of the sample. The biomasses of each of the sampled stump-root systems were calculated

based on the masses determined above and summed to obtain the total stand stump-root biomass (kg ha^{-1}). The aboveground biomasses for all the trees on the experimental plot were estimated with functions developed by Repola (2009).

3.2 Studies at the sprinkling infiltration site

The sprinkling infiltration study (study **III**) is part of a research project initiated by the Finnish Forest Research Institute (since 2015 Natural Resources Institute Finland, LUKE) at the end of the 1990s, which studied the effects of lake water infiltration on soil properties and processes, soil percolate water quality, tree growth and understory vegetation. The experiment was designed as a part of a sprinkling infiltration water plant, operated by the local waterworks. The findings of the experiments, conducted in 1998–2003 in the infiltrated stand, have been reported in peer-reviewed journals (Nöjd et al., 2009) and scientific and technical reports (Helmisaari et al., 2003; Derome et al., 2004; Derome et al., 2006; Helmisaari et al., 2006).

The effects of sprinkling infiltration on forest soil, understory vegetation and tree growth and their respective recovery were studied at an experimental site located in Vuontee (62°20′8′′ N, 26°2′5′′E), Central Finland (Figure 1). Water from a nearby lake was sprinkled directly onto the forest floor via a network of pipes (Figure 3). During the infiltration treatment, the amount of infiltrated water was 600 m^3 m^{-2}

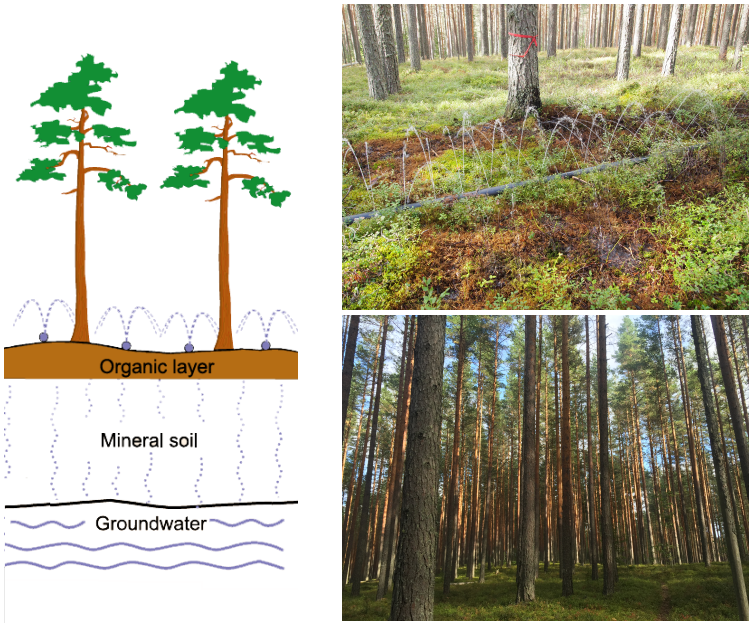

Figure 3 The mature Scots pine (*Pinus sylvestris*) stand located in Vuontee, Central Finland had been sprinkled with surface water, pumped from a nearby lake. Illustration adapted from Helmisaari et al. (2003) and Nöjd et al. (2009). (Photos: Lilli Kaarakka)

annually (ca. 600 000 – 1 000 000 mm), which is 1000 times higher than annual precipitation (643 mm yr^{-1}) in the stand (Nöjd et al., 2009). The infiltrated area was a mature (125–130 years) sub-xeric Scots pine-dominated, *Vaccinium vitis-idaea* (VT) site (Cajander, 1949).

The soil consists of relatively coarse sandy deposits and the soil type was identified as humic podzol. Prior to infiltration, few plant species dominated the understory; mosses such as *Pleurozium schreberi* and *Dicranum polysetum,* dwarf shrubs (*Vaccinium myrtillus* and *Calluna vulgaris)* and some lichens (*Cladina* spp). Four experimental plots (30 m x 30 m) were established in 1998; two of which were infiltrated during 20.9.1999–19.12.2001 (Helmisaari et al., 2003; Derome et al., 2004; Derome et al., 2006; Nöjd et al., 2009) and two remained as untreated controls until 2002. Parts of the control plots had been infiltrated 16.11.2002–2.5.2005, thus these were excluded from all the sampling in 2012–2015 for this current study and were replaced with near-by uninfiltrated plots. In order to ensure the comparability of the results, similar sampling methods were used in 2012–2015 as described above and in Derome et al. (2004) and Nöjd et al. (2009).

To study the effects of infiltration on soil N transformations and nutrient dynamics soil samples were collected in 2013 and 2014.The organic layer and mineral soil, to a depth of 40 cm, were sampled with a soil corer. In 2015, the stand understory plant species abundancy was visually estimated using a 1 m x 1 m quadrat-form-frame. Plant species of both field layer (grasses, forbs, shrubs, seedlings) and bottom layer (lichens and mosses) were identified and their percentage cover (0–100 %) was estimated. In the data analysis, species were organized into plant functional groups. Tree seedlings and shrubs under 0.5 m in height were included in the same group with dwarf shrubs. Field layer vegetation (grasses, herbs and forbs) formed one group and bottom layer vegetation, consisting of mosses, liverworts and lichens, was divided into two groups: mosses and lichens. The percent covers of all identified species within a functional group were summed up to give cover values for the functional groups. It is worth noting that to estimate plant community diversity Simpsons diversity index was used in study **III** and the indices inversion (Simpson's reciprocal) in study **II** in the calculations (*vegan*-package in R). Finally, in order to study tree radial growth, a total of 15 pines were cored in August 2013. Ten trees were cored on the infiltrated plots and five control pines outside the experimental area.

To homogenize the soil samples, the organic layer and mineral soil samples were milled. Total C and N concentrations were determined from the homogenized soil samples. Soil pH was measured with a glass electrode on suspensions of soil in demineralized water with a ratio of 20 mL sample and 50 mL of water (the samples had been left to stand overnight after mixing). The concentrations of exchangeable cations Al^{3+}, Ca^{2+}, Fe^{3+}, K^{+}, Mg^{2+} and Mn^{2+} were determined by ICP-OES following an extraction with barium chloride. Nitrogen transformations were studied in aerobic incubation experiments in the laboratory from fresh soil samples. The samples were

adjusted to 60 % water-holding capacity and incubated for 40 days in an constant temperature of +14 °C as described by Priha and Smolander (1999) and Paavolainen et al. (2000a). Net N mineralization and nitrification rates were calculated as described earlier (study **II**).

3.3 Statistical analyses

All the analyses for studies **I** and **II** were completed with SAS Ver. 9.3. (SAS Institute Inc., Cary, NC, USA). In studies **III** and **IV**, all the statistical analyses were completed using R version 3.4.2 (R Core Team, 2017).

Studies **I** and **II** were designed as block experiments with two treatments; whole-tree harvesting (WTH) and whole-tree harvesting combined with stump harvesting (WTH+S) in study **I**, and slash harvesting (S) and slash + stump harvesting (SS) in study **II**. The Haukilahti block consisted of three whole-tree harvested plots and three stump harvested plots, while the Honkola block consisted of one whole-tree harvested plot and one stump harvested plot. The trial was analyzed as a randomized block experiment with one treatment factor (stump harvesting) and with different degrees of disturbance, number of trees (individual species and total number of trees) and basal area of trees as response variables. Vegetation data (Simpson's reciprocal index, species richness, sub-plot beta diversity, PCA subplot scores) were calculated using the subplot as the principal observation unit. The effect of stump harvesting on soil C and N pools in different soil layers was tested with a mixed linear model, which took into consideration that the data for each soil layer, being on top of each other, is correlated. Depth was thus treated as a repeated factor in the analyses.

As for C and N mineralization rates (study **I**), different disturbance classes (Table 2) were analyzed separately. The statistical analysis was made as a split-plot ANOVA, in which treatment (WTH and WTH+S), soil layer (top, humus layer, 0–5, 5–10 and 10–20 cm mineral soil) and the interaction between treatment and soil layer were considered fixed factors, and where plot (within treatment) was a random factor. The differences between disturbance classes were analyzed according to a two-way ANOVA.

Due to operational limitations, only some variables were measured from both Honkola and Haukilahti (pH, C mineralization rate, C and N concentration, C/N ratio) (n = 4) whereas others were measured from Haukilahti (C pools, N pools and R_h = heterotrophic respiration) (n = 3). Net N mineralization and net nitrification data was only determined for the data from Honkola (n = 1). Consequently, treatment effects could not be evaluated at Honkola. However, by using the subplots at Honkola as replicates, it was possible to get an idea of the treatment effects assuming no plot differences.

In study **III**, t-tests were used to compare sprinkling infiltration effects on soil nutrient concentrations, N pool and transformations, soil pH and different plant

functional groups. In addition, t-tests were used for comparing treatment means for different plant functional groups (n=4) and treatments (n=2) sampled in 2015. Two-way repeated measures ANOVA was used for comparing treatment and the sampling years (1998, 2000, 2001, 2002, 2003) for different plant functional groups. Pairwise comparisons (using Tukey HSD) were used to find the difference between sampling years. Simpson's diversity index was used to estimate plant community diversity. Ordination was used to visually illustrate the differences in plant species composition between the infiltrated plots and the control plot. Prior to ordination the plant cover data was log-transformed to normalize it. Principal component analysis (PCA) was chosen as the ordination method to display the patterns of plant community differences. Two-way repeated measures ANOVA was used for comparing treatment and sampling years for different plant functional groups.

In study **IV**, prior to being analyzed with two-way ANOVA, surface disturbance proportions were log-transformed to obtain equal variances and normal distribution of the proportional data. Experimental site (time since harvesting), treatment and disturbance class and their interactions were defined as fixed factors. The differences between disturbance classes at each experimental site were analyzed according to a two-way ANOVA, where treatment, disturbance class and their interactions were considered as fixed factors.

4 RESULTS AND DISCUSSION

4.1 How stump harvesting affects soil C and N mineralization rates and pools?

The effects of stump harvesting on (i) soil C and N stocks, (ii) C mineralization and N transformations, (iii) the amount of CWD, (iv) soil surface disturbance and (v) vegetation (understory plants and seedlings) were studied at Haukilahti and Honkola, in Central and Southern Finland 8–13 years after final-felling (Table 1). Logging residues were harvested in all stands and the stands were planted with Norway spruce seedlings after site preparation in the form of mounding.

Based on previous studies, we hypothesized that stump harvesting increases soil disturbance and the treatment effect would be most pronounced in the organic layer, where we expected to observe a decline in C and N pools (study **II**). Following stump harvesting, soil C and N pools tended to be lower, however this difference was not statistically significant due to the high variation between plots. Organic layer C pool was estimated to 22 Mg ha^{-1} and 13 Mg ha^{-1} following mounding and stump harvesting (data not shown). Corresponding values for mineral soil (down to 25 cm) were 8.1 Mg ha^{-1} and 7.5 Mg ha^{-1}, respectively. N pools in the organic layer were 785 kg ha^{-1} and 561 kg ha^{-1}, and 389 kg ha^{-1} and 391 kg ha^{-1} in the mineral soil for the mounded sites and stump harvested sites, respectively. From the two study sites, Honkola was more fertile and some of the stands had a thicker organic layer than Haukilahti but the differences in C pools between the experimental sites in the mineral soil were minor.

In addition to total soil C and N pools, pools for each soil disturbance class (undisturbed soil, mounds and pits) and soil layer were estimated for samples from Haukilahti (study **I**). No statistically significant differences between stump harvested and mounded sites in terms of soil total C and N pools were detected. Separated per disturbance class, total C and N amounts for the whole soil profile were significantly higher ($p \leq 0.05$) in the mounds and undisturbed surfaces than in the pits (Figure 4). Total soil C pools were estimated to be 69 and 56 Mg ha^{-1} in the mounding and stump harvesting treatments in the undisturbed soil, respectively (to depth of 20 cm) (Figure 4). Corresponding total C pools for the mounds and pits were 80 and 73 Mg ha^{-1} and 18.5 and 23 Mg ha^{-1}, respectively (Figure 4). In the undisturbed soil, total soil N pools were 3200 and 2600 kg ha^{-1} (to depth of 20 cm) in the mineral soil, 3480 and 3570 kg ha^{-1} in the mounds and 1084 and 1382 kg ha^{-1} in the pits in the mounded and stump harvested sites, respectively. C concentrations were significantly lower at the stump harvested sites compared to mounded sites in the mound disturbance class, whereas C and N concentrations in other disturbance classes only showed differences between soil layers (data not shown). The organic layer had higher C and N concentrations than any of the other soil layers, including the top layer, which consisted of a mixture of organic and mineral soil. The tendency of C decrease was restricted to the organic layer, both in the mound and in the undisturbed classes, which in part confirmed one of our hypotheses.

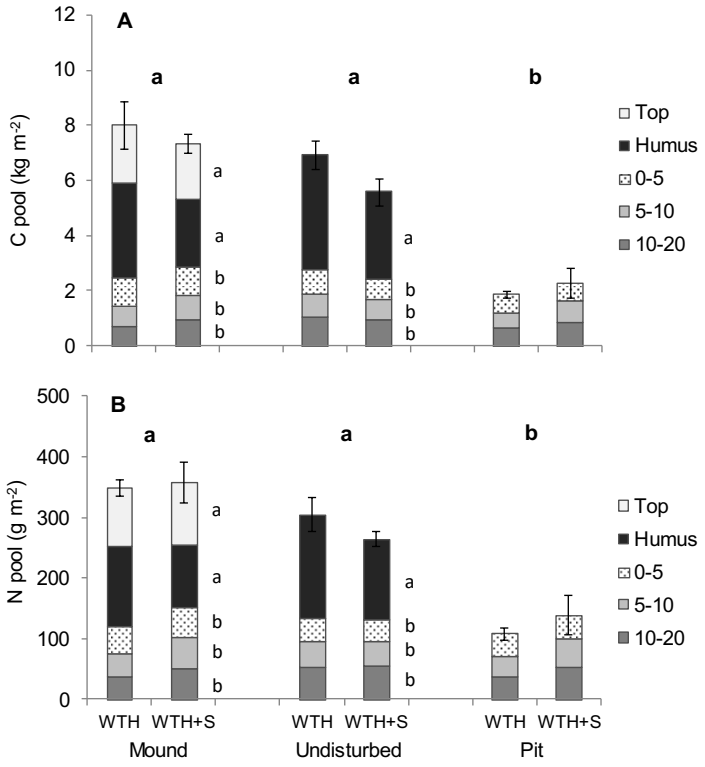

Figure 4 Mean soil C (A) and N (B) pools (± SE, for the whole soil profile) at Haukilahti (n=3) in two different treatments (WTH = whole-tree harvest; WTH+S = removal of logging residues and stumps), three disturbance classes (mounds, undisturbed soil and pits) and five soil layers (top=inverted humus + upper mineral soil layer, humus layer, 0–5, 5–10 and 10–20 cm soil layers). Different bold letters denote significant differences ($p \leq 0.05$) between the disturbance classes (independent of treatments) and plain letters denote significant differences between the soil layers (independent of treatments) in the same disturbance class. There were no significant differences between the treatments.

C/N ratios had a tendency to be lower after stump harvesting ($p = 0.08$ and $p = 0.11$) for the mound and undisturbed classes, respectively (study **I**). This might indicate that SOM from deeper soil layers had been intermixed with the more superficial soil layers, because C/N ratios normally decrease with increasing soil depths in boreal forests (e.g. Callesen et al., 2007). Similar results have been reported in Finland by Kataja-aho et al. (2012b), who found lower C/N ratios post-stump harvesting, and accounted it to the increased mixing caused by harvesting. The C concentration was significantly reduced by stump harvesting in the mound disturbance class ($p \leq 0.05$), which could also be

explained by both soil mixing and increased decomposition of SOM. Soil pH was also higher in organic layer following stump harvesting ($p = 0.001$) (studies **I** and **II**).

In addition, the objective was to determine if stump harvesting causes qualitative and quantitative changes in the SOM stock in different soil-surface disturbance categories (study **I**). Our specific hypothesis was that mixing of the soil layers (incorporation of OM into the mineral soil and *vice versa*) caused by stump harvesting will increase decomposition and will therefore increase C and N mineralization rates in topsoil layers (per unit C). Furthermore, the highest fluxes of CO_2 were expected from the surfaces with high SOM content (e.g. the mounds). The excavation pits – lacking the organic layer entirely – were expected to produce the lowest CO_2 fluxes. Contrary to our hypothesis, there was no significant effect of stump harvesting on soil C mineralization rates (Figure 5) (study **I)**. The high C mineralization rate measured from the pit-soil samples was unexpected, however. It is possible that the pits acted as collectors of fresh litter from the clear-cut vegetation, and although all visible litter was removed at sampling, the build-up of fragmented litter (during the time since harvesting; 11–12 years) might have created an influx of high quality SOM to the top layer in the pit. It is also possible that the pits are filled with water following heavy rains and snowmelt. During litter decomposition, fragmented litter and dissolved organic C (DOC) was probably formed, and could penetrate into the underlying soil layer and act as a C source for the decomposers. The early dynamics of CO_2 efflux from soil pits was studied by Pumpanen et al. (2004), who reported low soil CO_2 efflux from exposed E and C horizons during the first year after clear-cutting, followed by a steady increase in CO_2 efflux during the following two years which they suggested was caused by vegetation colonizing the pit.

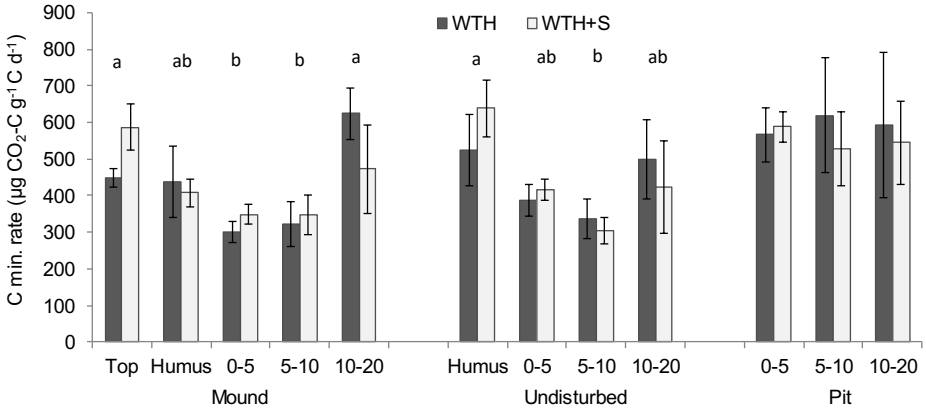

Figure 5 Mean C mineralization rate (±SE, n=4) in the two different treatments (WTH and WTH+S), three disturbance classes (mounds, undisturbed soil and pits) and five soil layers at Honkola and Haukilahti. There were no significant differences between the treatments. Detailed explanations of the figure features are given in Figure 4.

In our study, the top layer of the mounds contained extra SOM: an inverted humus layer plus attached mineral soil on top of the original humus layer (study **I**). This top layer seemed to have similar C mineralization rates as the underlying humus layer (Figure 5). In a study conducted 1–2 years after stump harvesting, Mjöfors et al. (2015) observed higher CO_2 fluxes from the plots with double humus layer than from the control plots with an undisturbed humus layer. However, during the second year, the soil CO_2 fluxes from the plots with a double humus layer decreased and equaled those from control plots (Mjöfors et al., 2015). Thus, Mjöfors et al. (2015) concluded that although mixing favors decomposition it does not necessarily lead to higher CO_2-emissions from the whole profile, but rather creates SOM cohorts in various soil depths, which then decompose at different rates.

Annual heterotrophic respiration (R_h) extrapolated to field conditions was estimated to range between 1000 and 1400 g CO_2-C m^{-2} annually in the undisturbed and mound soil profiles (Figure 6) (study **I**). R_H for the whole soil profile did not differ significantly between stump harvesting and mounding, but in the mound disturbance class, R_H was significantly lower in the organic layer of the stump harvested stand. The top and humus layers had significantly higher R_H than the mineral soil layers. Estimated R_H for the whole soil profile in the mounds (mean 13 Mg CO_2-C ha^{-1} yr^{-1}) and undisturbed soil (mean 12 Mg CO_2-C ha^{-1} yr^{-1}) was significantly higher than in the pits (mean 4.5 Mg CO_2-C ha^{-1} yr^{-1}).

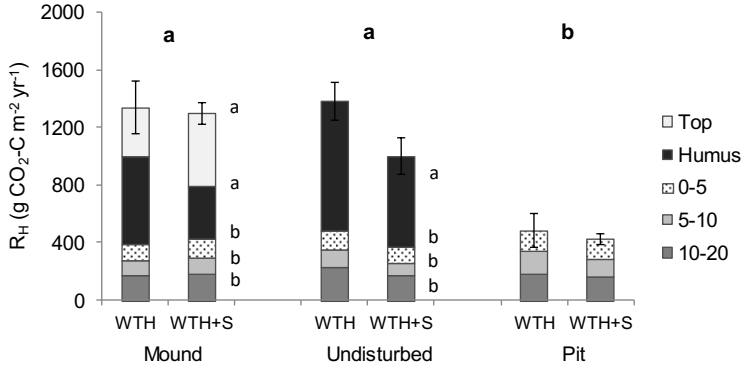

Figure 6 Mean annual heterotrophic respiration (R_h) in two different treatments (WTH and WTH+S), three disturbance classes (mounds, undisturbed soil and pits) and five soil layers at Haukilahti (n=3) extrapolated to the field but with the same temperature (15 ºC) and soil moisture (60% WHC) conditions over the whole year as measured in the laboratory. SE (n=3) given for the whole soil profile. Different bold letters denote significant differences ($p \leq 0.05$) between disturbance classes (independent of treatments) and plain letters denote significant differences between soil layers (independent of treatments) in the same disturbance class. The only treatment effect was found in the mound humus layer, where R_h was significantly lower in the WTH + S than in the WTH treatment.

N transformations were only studied at Honkola (study **I**). Net N mineralization (µg gC^{-1}) rate obtained in the laboratory did not show clear differences between the treatments and net nitrification was negligible (Figure 7). Stump harvesting had a tendency (p = 0.053) to increase net N mineralization in the pits however (Figure 7). In Finland, Kataja-aho et al. (2012b) studied exposed mineral soil surfaces post-mounding and stump harvesting and found that net N mineralization was initially higher on stump harvested sites. Yet the differences in net N mineralization and net nitrification rate between treatments diminished by year five since harvest (Kataja-aho et al., 2012b). They also observed a higher CO_2 flux from stump harvested sites in the field, however this effect was not observed in the *in vitro* samples (Kataja-aho et al., 2012b).

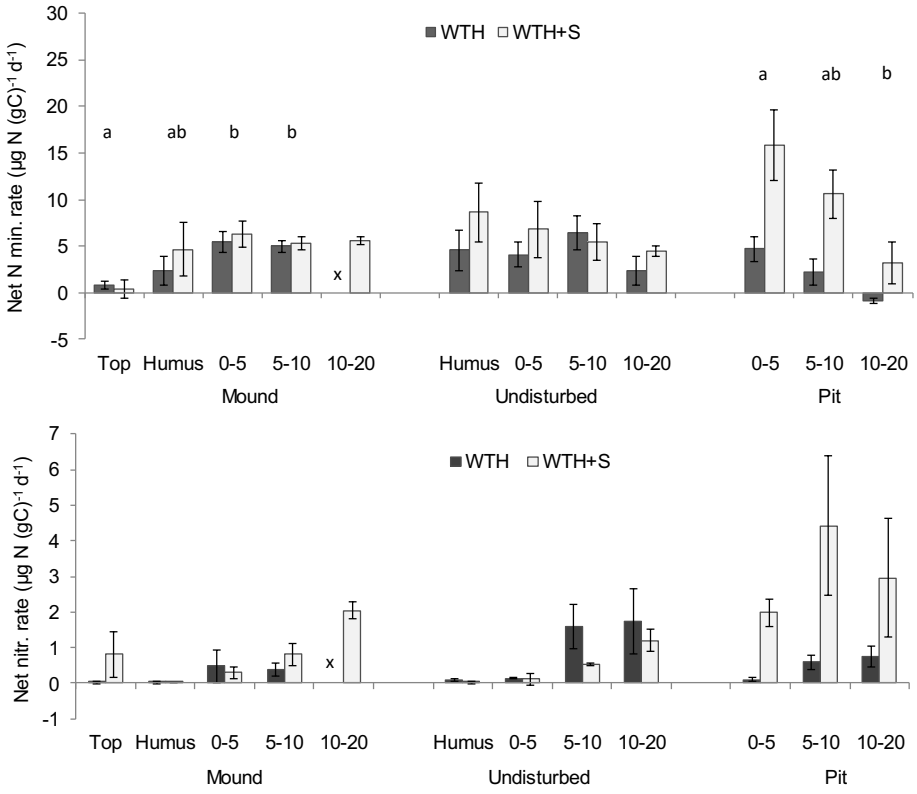

Figure 7 Net N mineralization rate and nitrification (± SE) in two different treatments (WTH and WTH+S), three disturbance classes (mounds, undisturbed soil and pits) and five soil mineral layers at Honkola. SE based on three subplots within two un-replicated plots. Different letters denote a significant difference ($p \leq 0.05$) between soil layers in the same disturbance class. No significant differences between the treatments were found, but stump harvesting had a tendency (p = 0.053) to increase net N mineralization in the pits. x = no data.

A recent study by Persson et al. (2017) reported significantly lower C pools in the organic layer and a tendency towards higher pools in mineral soil (0–10 cm) 20–30 years after stump harvesting when compared to mounding, but no effect on soil C mineralization nor N transformations nor soil N pools. They also reported a reduction in heterotrophic respiration (R_h), which they conclude was the result of the lower C pool, as C mineralization rate was not significantly affected by stump harvesting (Persson et al., 2017). This study was conducted in eight Swedish stands, that prior to clear-cutting had been mixed conifer stands (Scots pine and Norway spruce) and in contrast to our study sites, following harvesting all but one stand had been planted with Scots pine seedlings. In our study (study **I**), the organic layer had lower C concentration, lower C/N ratio and higher pH after stump removal, all indicating that the organic layer had partly been mixed with materials from the other soil layers during stump harvesting.

Only a handful of studies have attempted to quantify how stump harvesting affects soil C and N pools in the long-term. Karlsson and Tamminen (2013) found no treatment effects on soil C and N pools 30 years after stump harvesting, while Zabowski et al. (2008) reported decreased C and N stores 22–29 years after stump removal. Strömgren et al. (2013) found a reduction in the organic layer C stock when stump harvesting was combined with logging residue harvesting, but reported that the effects were site-specific. In Canada, Hope et al. (2007) found no difference in SOC stocks in the organic layer nor in the mineral soil between a stump harvested site and an untreated control, 10 years after treatment. In Sweden, 32–39 years since harvesting, Jurevics et al. (2016) reported no effects of stump harvesting on soil C and N pools, nor tree biomass C but rather an insignificant species- and site-interaction in terms of treatment response. In a study by Egnell at al. (2015) soil C and N pools were reduced, whereas seedling growth and survival was improved by stump harvesting and deep soil cultivation, in which the organic layer was displaced 50–60 cm deep into the mineral soil, 22–24 years following harvesting. However due to the high intensity of the deep soil cultivation treatment, this result is not entirely applicable in current forest management practices.

4.2 Surface disturbance and vegetation dynamics following stump harvesting

The effects of stump harvesting on soil surface disturbance were studied at all stump harvesting sites in two separate sets (Table 1); in study **II** at Honkola and Haukilahti (2013) and in study **IV** at Haukilahti, Hyvinkää and Karkkila (2014/2015). At Haukilahti and Honkola, 56 % of soil was considered disturbed post-stump harvesting and 43 % after mounding, however this difference was not statistically significant (study **II**). The increase in exposed mineral soil after stump removal was the most pronounced type of disturbance (study **II**).

In study **IV**, the three experimental sites all represented a different age class (i.e. time since harvesting); old (Haukilahti), mid (Karkkila) and young (Hyvinkää). It was

evident that more undisturbed surface (70–73 %) had remained at the mounded sites, compared to the sites where the stumps had been harvested (40–54 %) (Figure 8). However, neither stump harvesting nor time since harvesting were significant factors in explaining the difference in the proportion of different soil disturbance classes. Only at the youngest site (Hyvinkää) there were significantly more mounds at the stump harvested sites ($p = 0.027$). The findings of this study are consistent with other studies, which have estimated that more soil surface is disturbed at stump harvesting compared to mounding (Strandström, 2006; Kataja-aho et al., 2011b; Strömgren and Mjöfors, 2012; Berg et al., 2015; Tarvainen et al., 2015; Strömgren et al., 2017). However, to our knowledge, this study is the first to assess the development of harvest-induced soil surface disturbance dynamics across a temporal gradient (i.e. time since stump harvesting). The results of this study indicate that soil disturbance caused by stump harvesting exist for a long period of time. More than half of the soil surface was considered disturbed after stump harvesting at the most mature site (Haukilahti) logged 13 years ago, whereas two thirds remained undisturbed at the mounded plots (study **IV**).

Because our study sites had similar soil texture, climatic conditions, vegetation and silvicultural methods, the differences resulting from site variation were probably small. Thus, it seems evident that stump harvesting combined with mounding causes a greater soil surface disturbance than mounding alone. In this study (**IV**), wheel ruts were not

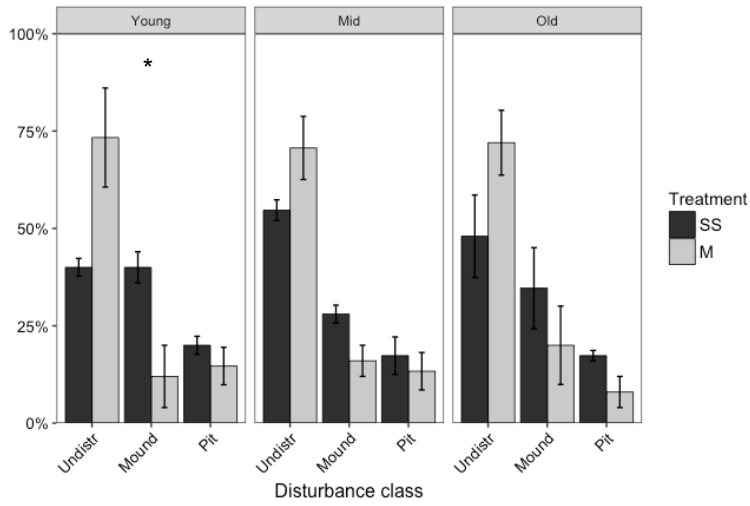

Figure 8 Proportions of the different surface disturbance classes (undisturbed, mound and pit) (%, ± SE) at the different Norway spruce stands clear-cut in 2010 (young – Hyvinkää), 2007 (mid – Karkkila) and 2001 (old – Haukilahti). The two treatments were stump harvesting combined with mounding (SS) and mounding (M) (n=3). Asterisks indicate statistical significance for the difference of the treatment means ($p \leq 0.05$).

included in the disturbance classification as they were either absent or considered a minor disturbance in the experimental plots. At all the experimental sites, mounding had been carried out immediately after stump harvesting, thus it is possible that possible wheel ruts had been covered by that treatment.

Understory vegetation dynamics following stump harvesting were studied at Haukilahti and Honkola (study **II**). Stump harvesting had no significant effect on the cover of understory plant species, total moss cover or vegetation composition. However, the coverage of *Vaccinium myrtillus* and *Vaccinium vitis-idaea* tended to be lower on stump harvested plots while the cover of *Rubus idaeus* tended to be higher. Total stem density was almost twice as high in stump-harvested stands as in stands where stumps were not harvested, due to significantly higher stem density of *Betula pendula* Roth. The positive response of *Betula* spp. to stump removal could be explained by the increased germination on disturbed, exposed mineral soil (study **II**). This observation is in-line with the findings of Saksa (2013) who reported that the number of birch seedlings was higher after stump harvesting as a result of the higher abundance of disturbed soil surfaces. Karlsson and Tamminen (2013) also reported that stump harvesting improved the survival of planted trees (Norway spruce and Scots pine) and increased the natural regeneration of deciduous trees. Kataja-aho et al. (2011b) described that the exposed mineral soil as a result of stump harvesting benefitted plant cover and thus increased plant diversity and species richness. Following forest harvesting it is possible that re-emerging vegetation acts as a transient sink for nutrients thus limiting the losses to leaching (Palviainen et al., 2005a). Bergholm et al. (2015) reported that clear-cutting resulted in marked accumulation of inorganic N (NH_4–N) in soil during the first year after clear-cutting, and the accumulation of inorganic N in the soil was higher than plant uptake or leaching. Interestingly, stump and dead root immobilization of N was thus quantitatively almost as important as the other two pathways; plant uptake and leaching (Bergholm et al., 2015).

However, most of the previous studies in the Nordic countries on the effects of stump harvesting on ground vegetation have been restricted to the first few years following harvesting (Kataja-aho et al., 2011a; Tarvainen et al., 2015). The period covered in study **II**, 8–13 years, possibly represents a time period in stand succession where the immediate effects of clear-cutting and soil disturbance may have decreased or – in part – vanished. A study conducted in south-central Sweden in stands that had been harvested 1977–1989 concluded that stump harvesting did not have long-term effects on understory vegetation (Rudolphi and Strengbom, 2016). It is also possible that the effects of stump harvesting are masked by the disturbance-effect caused by clear-cutting and site preparation (Tarvainen et al., 2015; Rudolphi and Strengbom, 2016). In the US, Kaye et al. (2008) reported that stump removal substantially changed the forest understory species composition favoring early-successional forbs and grasses over slow-growing shrubs. Nevertheless, the findings of Tarvainen et al. (2015) underscore

the high site-dependency of the effects of stump removal on ecological variables, such as vegetation.

4.3 Biomass distribution of harvested stump-root systems

Given that large quantities of biomass are removed from the stand in stump harvesting (e.g. Eräjää et al., 2010) it was expected that stump harvesting results in reduction of CWD at the study sites (studies **II** and **IV**). At Haukilahti and Honkola considerable amounts of C were removed by stump harvesting; 7.3 Mg C ha^{-1} in the aboveground part of stumps and 28 Mg C ha^{-1} when coarse roots were also included in the calculations (data not shown). The largest amount of C, 16 Mg ha^{-1}, was removed with the coarse roots (diameter > 5 cm) (study **II**). The amount of remaining stump and root biomass was significantly lower in stump-harvested stands than in stands where stumps had been retained ($p \leq 0.05$), whereas differences between mounding and stump harvesting were not significant for the other types of CWD (study **II**). Correspondingly, in a study carried out in southern Central Finland, Rabinowitch-Jokinen and Vanha-Majamaa (2010) reported that stump harvesting combined with logging residue harvesting (and mounding) significantly decreased the total volume of aboveground CWD. Apart from the lower number of stumps at stump harvested experimental plots, average diameter of the remaining stumps was also smaller (study **II**). This finding is in-line with Eräjää et al. (2010) who reported that due to operational bias, larger stumps are more frequently extracted and smaller stumps (diameter < 19 cm) remain in the stand post-harvesting. From a bioenergy perspective bigger stumps come with an added yield, bearing in mind however that the average soil area disturbed increases exponentially with increasing stump size (Berg et al., 2015).

At Karkkila where entire stump and root systems were excavated (study **IV**), the total stump and coarse root biomass was 39 300 kg ha^{-1} and 79 % (30 979 kg ha^{-1}) of this biomass was removed during stump harvesting and 21 % (8 340 kg ha^{-1}) remained in soil (Table 3). The stump-root system accounted for 17 % of the whole-tree biomass. Coarse roots and fine coarse roots together represented the largest biomass component in the stump-root system, accounting for 73 % of the total stump-root system biomass and 12.1 % of the whole tree biomass. The N stock in the stem and stump-root biomass was estimated to 202 kg ha^{-1} (data not shown). This excludes needles, branches and fine-roots. The tree stem, including both wood and bark, was the largest N pool at 158 kg ha^{-1}, accounting for 78 % of the harvested N. Correspondingly, the stump-root system had a stock of 44 kg N ha^{-1}. Bark was a significant pool of N, both in the tree stem and the stump-root system; bark accounted for 40 %, 27 % and 42 % of the total N in the stem, stump and coarse roots, respectively.

Table 3 Distribution of biomass (kg ha^{-1} dry-weight) in stumps, coarse roots (diameter > 35 mm), fine coarse roots (diameter = 5–35 mm) and the stem and the total aboveground tree biomass at the Karkkila experimental site (33 trees). Distribution of biomasses as percentages are indicated in italic.

		kg ha^{-1}	% total tree biomass	Wood density (incl. bark) kg m^3
Total aboveground[1]		196 241	83	
Stem[2]		162 636	*69*	
Total stump and roots	% of stump-root system	39 319	*17*	
Stump[2]	*27*	10 489	*4.5*	361
Coarse roots[2]	*61*	23 984	*10*	398–441
Thin coarse roots[2]	*12*	4 846	*2.1*	452
	Total aboveground + stump-root biomass (kg ha^{-1})	235 561	100	

[1] (including bark, needles and branches)
[2] (including wood and bark)

Coarse root wood density tended to have an inverse relationship with root diameter; density increased in the finer roots (Table 3). Anchoring the tree is one of the most important functions of tree roots (Kalliokoski et al., 2008) and most trees produce dense wood at the stem base (i.e. stump) (Hakkila, 1989). However, growth rate tends to be faster in the roots near the stump, resulting in larger radial growth and lower wood density (Hakkila, 1989). Kalliokoski et al. (2008) concluded that root-systems' plasticity results in large variation between tree individuals in terms of root-system size and shape. Roots have been assumed to decompose faster than stumps due to their smaller size (i.e. diameter) and higher moisture content, however studies using a chronosequence of different aged Norway spruce stands, have reported that stumps decay significantly faster (Palviainen and Finér, 2015) or at the same rate with the roots (Shorohova et al., 2012). In contrast to coarse roots, a majority of the understory vegetation (herbs, forbs, grasses and dwarf shrubs) and tree fine roots are in the organic layer or in the first 30 cm of the mineral soil (Helmisaari et al., 2007). Therefore, although stumps and coarse roots contain small amounts of nutrients compared to logging residues and fine roots (Palviainen and Finér, 2015) they potentially have a disproportionate importance in the C and N dynamics deeper in the mineral soil, outside the realm of finer, nutrient rich roots.

4.4 Sprinkling infiltration and ecosystem recovery

The effects of chemical disturbance on forest soil were studied on an esker located in Vuontee, Central Finland (study **III**). The Scots pine stand had been sprinkled with lake water to produce artificial groundwater in the end of 1990s.

4.4.1 Soil acidity and nutrient status

In this study (study **III**), we found that pH had remained significantly higher on the infiltrated plots over a decade after infiltration had stopped, particularly in the mineral soil (Figure 9). During the infiltration in 1999–2001, soil pH rose significantly at the infiltrated plots (Nöjd et al., 2009). Previous studies conducted in the same experimental stand (Helmisaari et al., 2003; Derome et al., 2004; Nöjd et al., 2009) reported an increase from pH 3.8 to 6.7 in two years. Similar results have been reported at other sprinkling infiltration sites (Paavolainen et al., 2000a; Lindroos et al., 2001).

No significant effect of infiltration on soil C and N concentrations was detected (study **III**). Also, no statistically significant difference were observed in the C/N ratio of the soil (data not shown). OM % in organic layer tended to be higher at the control plots than at the infiltrated plots, 59 % and 46 %, respectively (data not shown).

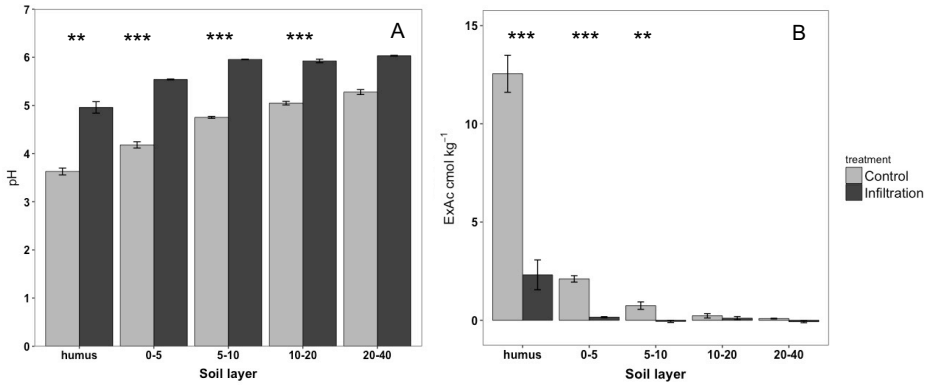

Figure 9 Soil pH (A) and exchangeable acidity (B) (cmol kg^{-1}) (± SE) for organic layer (humus) and mineral soil layers (cm) for the two different treatments in 2013, 12 years after the infiltration had stopped at the infiltration experimental site . Infiltration plots had a significantly higher pH compared to the control plots in all soil layers, except in the deepest mineral soil layer. Exchangeable acidity was significantly higher at the control plots in the humus layer and the first 10 cm of the mineral soil. Asterisks indicate statistical significance for the difference of the treatment means ($p \leq 0.05$*, $p \leq 0.01$**, $p \leq 0.001$***).

Concentrations of exchangeable Ca^{2+}, K^{+}, Mg^{2+} and Al^{3+} (mg kg^{-1} dry weight) were significantly different between the treatments, particularly in the deeper layers of the soil. Ca^{2+} and Mg^{2+} concentrations were significantly higher on infiltrated plots than at the control plots in 0–5 and 10–40 cm of the mineral soil (study **III**). In contrast, exchangeable Al^{3+} concentrations were much higher at the control plots. Furthermore, exchangeable acidity was significantly lower at the infiltrated plots thus implying that the acid cations had been largely replaced with base cations, Ca^{2+} and Mg^{2+}, during infiltration (Figure 9). At the control plots in contrast, the concentration of Al^{3+} and exchangeable acidity were both higher, thus indicating that the exchange sites were controlled more by Al^{3+} than by base cations. The amount of free Al^{3+}, and Al^{3+} bound at the exchange sites or to the organic matter complexes is closely related to the soil pH. These results are in agreement with the findings of Lindroos et al. (2001) and Nöjd et al. (2009) who found that the saturation of the cation exchange sites by base cations occurred soon after the start of the infiltration. Furthermore, Nöjd et al. (2009) found no decline in Ca^{2+} and Mg^{2+} two years after the termination of infiltration. It appears that the hydrogen ions and Al^{3+} were relatively quickly displaced from the soil cation exchange sites by Ca^{2+} and Mg^{2+} during the infiltration. All these findings indicate that sprinkling infiltration has long-term effects on soil acidity.

4.4.2 Effects of sprinkling infiltration on soil N transformations

Changes in NH_4-N and NO_3-N concentrations were detected at the infiltrated plots, 13 years after infiltration (study **III**). The rate of net N mineralization (per SOM) was significantly higher at the infiltrated plots both in the organic layer and in the mineral soil (Figure 10). Also, NH_4-N concentration (per SOM) was higher in the organic layer of the infiltrated plots. Some net nitrification occurred in the mineral soil at the infiltration plots, but the rate was very low as was also the NO_3-N concentration (Figure 10). In the mineral soil, NO_3-N concentrations had increased significantly at the end of the infiltration in 2001 (Nöjd et al., 2009). Previous studies on more fertile sites have shown that infiltration initiates nitrification (Paavolainen et al., 2000a; Paavolainen et al., 2000b; Nöjd et al., 2009). Interestingly, this effect had previously only been observed in the mineral soil (Paavolainen et al., 2000a; Nöjd et al., 2009) and this increase in nitrification was probably very much linked to the higher soil pH. Paavolainen et al. (2000a; 2000b) reported that when excess of NH_4-N was present in the soil, soil pH was the major factor in determining the rate of NO_3-N production and concluded that the nitrifiers at the infiltrated plots were more sensitive to changes in pH and adapted to a higher pH than their counterparts at the control plots. The study by Paavolainen et al. (2000b) was carried out in a highly fertile sitein Southern Finland, where nitrification had remained high two-years after the termination of the infiltration.

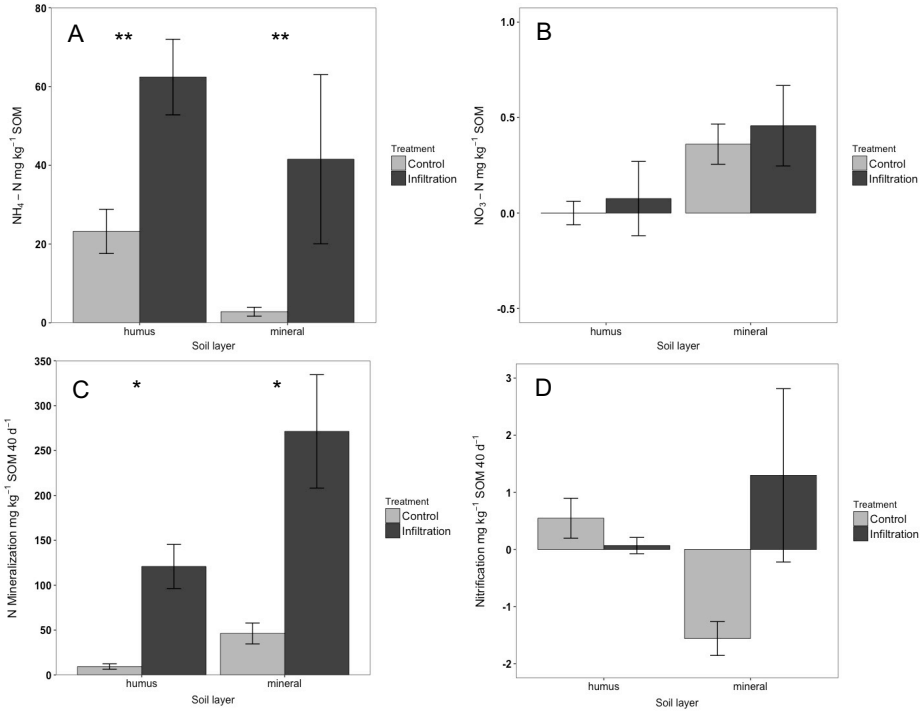

Figure 10 NH_4-N (A) and NO_3-N (B) (± SE) concentrations and rates of net N mineralization (C) and net nitrification mg kg^{-1} (D) (+/– SE) for the organic layer (humus) and mineral soil (0–10 cm) in 2014, 13 years after the infiltration had stopped. All values are per soil organic matter (SOM). The rate of net N mineralization and NH_4-N were significantly higher at the infiltrated plots. Asterisks indicate statistical significance for the difference of the treatment means ($p \leq 0.05$*, $p \leq 0.01$**, $p \leq 0.001$***).

Nitrogen addition, soil pH and NH_4-N availability are amongst the environmental factors which control and alter the composition, abundance and diversity of the nitrifying microorganism community (ammonia-oxidizing bacteria and archaea) in the soil (Levy-Booth et al., 2014). So why was net nitrification in the organic layer at the infiltrated plots low even though the conditions in the soil seemed to be ideal at the experimental site (study **III**)? Low nitrate concentration could be explained by the uptake of nitrate by the understory vegetation but this does not explain the observed low net nitrification. Contrary to intensive nitrification on more fertile sites due to infiltration (Paavolainen et al., 2000a), nitrification did not dominate at our study site in such a high rate, maybe due to the low fertility of the site. One possible explanation for the net nitrification occurring predominantly in the mineral soil could be the lack of

inhibitory organic compounds, such as terpenes, occurring in high concentrations in the organic layer (Smolander et al., 2012).

4.4.3 Vegetation responses to infiltration

No statistically significant differences between the treatments for different plant functional groups (mosses, shrubs, lichens, grasses) were found for the plant survey completed in 2015 (Figure 11) (study **III**). The total number of species tallied at the control and infiltrated plots were 14 and 22, respectively. In this study, there was a statistically significant ($p = 0.039$) difference between understory vegetation Simpson's diversity indices for the two different treatments (control 0.7, infiltration 0.62). Simpson's diversity index takes into account both species richness and evenness of the abundance of the species. The control plots had higher diversity indices, however a species tally did reveal that the infiltration plots had higher individual plant species richness (i.e. number of species), thus, it was likely that the lower species evenness was reflected to the Simpson's index.

Finally, the control plots had more litter (including woody and leaf litter) than the infiltrated plots at the time of 2015 sampling, 35 % and 30 % respectively (data not shown). The different plant survey years and the cover of each plant functional groups were also compared in terms of the effect of infiltration. For mosses, there was a significant treatment and year interaction, and pairwise comparisons revealed a difference in abundance in 2001, 2002 and 2003 ($p = 0.005$, $p = 0.002$, $p = 0.002$,

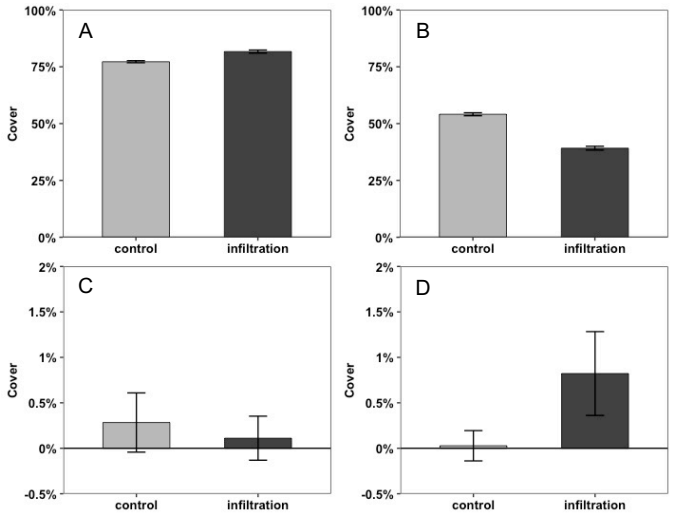

Figure 11 Cover of the plant functional groups (%, ± SE): mosses (A), dwarf shrubs (B), lichens (C) and forbs and grasses (D), 14 years after the termination on infiltration in 2015 at the control plots (light grey) and the infiltrated plot (dark grey).

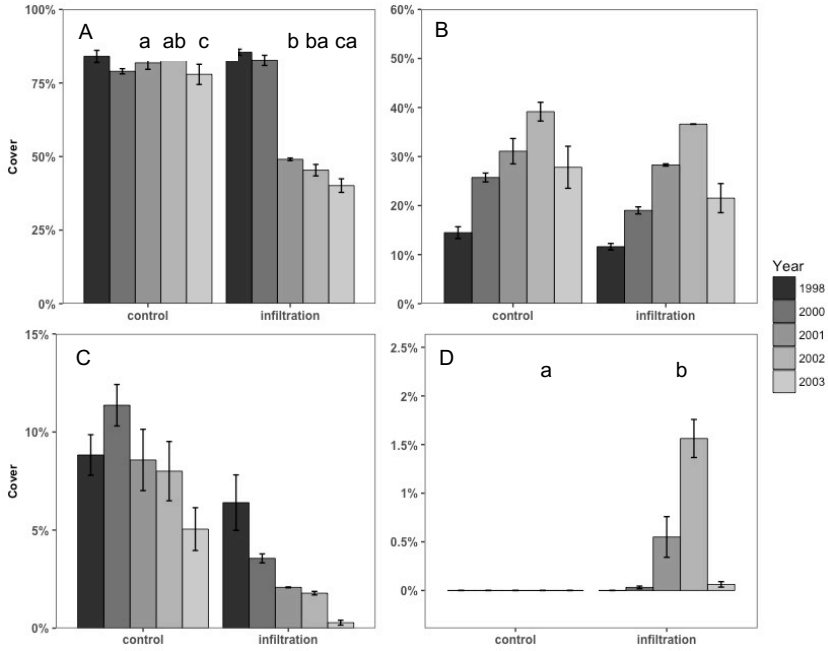

Figure 12 Cover of the plant functional groups (%): mosses (A), dwarf shrubs (B), lichens (C) and herbs and grasses (D) (± SE) for different years; before (1998), during (2000–2001) and after (2002–2003) the infiltration. Different letters indicate a significant difference between the treatments ($p \leq 0.05$).

Figure 12). Similarly, for grasses, there was a significant difference between treatments in 2002 ($p = 0.006$, Figure 12). Small shrubs, tree seedlings, and lichens showed an overall treatment effect on the abundance of these species but pairwise comparisons between the years did not reveal a difference.

Due to the infiltration treatment on a part (approximately 20–30 %) of the previous control plots, the results from infiltrated plots were compared with the results of the trees outside of the experimental area. Prior to the infiltration treatment, the pattern of radial growth was similar between the sample trees from the infiltrated plots and the control (Figure 13). During and after the infiltration in 1999–2001 tree growth doubled on the infiltration plots but remained stable in the control trees. At the time of the sampling in 2013 there was no difference between the ring width of the control trees and the trees at the infiltrated plots. Previous experiments at the our experimental site have reported a reduction in the cover of lichens and mosses between 1998–2003, and an emergence of new herbs and grasses, such as *Epilobium* sp. and *Tussilago farfara*, and finally, a gradual increase in the cover of dwarf shrubs at all plots, irrespective of the treatment (Derome et al., 2004). In the survey carried out in 2015, it became apparent that the cover of mosses had recovered from the infiltration and the abundance

of dwarf shrubs had remained high at the infiltrated plots. Herbs and grasses too were still more abundant at the infiltrated plots. In contrast, cover of lichens had reduced at all plots irrespective of the treatment. The added moisture and standing water, together with competition from the herbs, likely contributed to the sharper decrease in the cover of lichens at the infiltrated plots during infiltration. In addition, an increase in crown cover may have favored the recovery of shade-tolerant mosses and resulted in the reduction of light-demanding lichens.

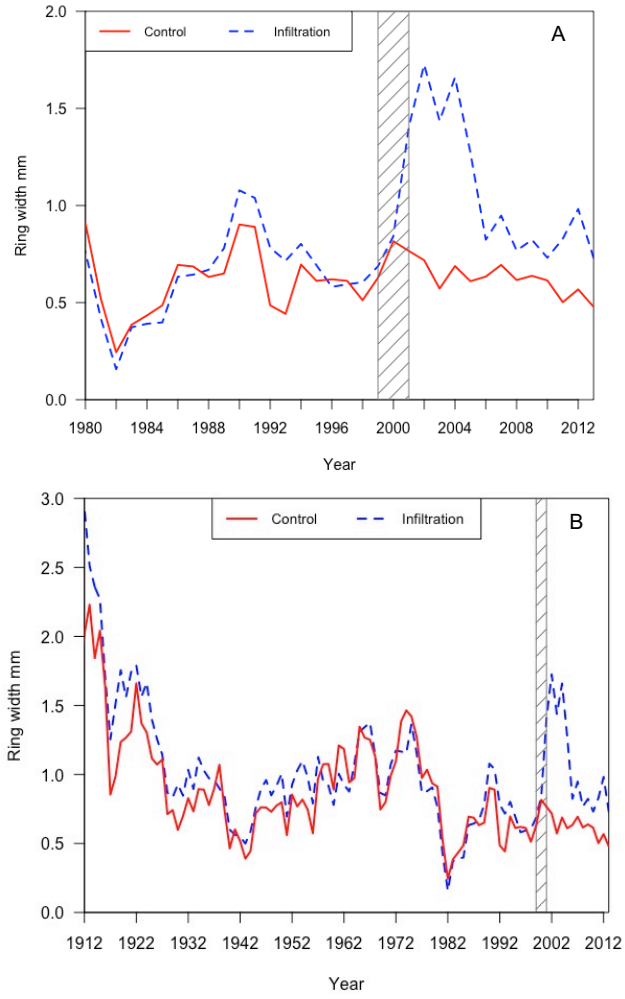

Figure 13 Mean annual radial growth of Scots pines in the infiltrated area (blue dotted line) and in the control plots (red solid line). The time period shown is 1980–2013 (A) and 1912–2013 (B). The highlighted grey area indicates the period of infiltration (1999–2001).

The changes in the vegetation composition caused by sprinkling infiltration are probably a result of both the altered moisture conditions and N added in the infiltration water. Sprinkling infiltration entails a certain dichotomy; as a treatment it is both a form of fertilizing and due to the large quantities of water added also a disturbance. Nitrogen fertilization and disturbance seem to have similar effects on understory vegetation, however different species can be more susceptible to either-or, or both (Saarsalmi and Mälkönen, 2001; Manninen et al., 2009; Metcalfe et al., 2013). Repeated disturbances have been reported to decrease the total understory biomass and reduce the microbial C pool (Manninen et al., 2009). Manninen *et al.* (2009) reported dwarf shrubs such as *Vaccinium myrtillus* and *Vaccinium vitis-idea* being more susceptible to physical disturbance than to N increase (fertilization or deposition). In Finland, forest N fertilization have been observed to result in an increase in the cover of grasses and some early-successional herbs such as *Rubus idaeus* (Saarsalmi and Mälkönen, 2001). Mosses, in contrast, have been reported to respond negatively to N fertilization (Saarsalmi and Mälkönen, 2001) and large scale disturbance, such as logging (Tarvainen et al., 2015; Tonteri et al., 2016). Fertilization effects on tree growth have been studied in the Nordic region. Tree growth increases following N fertilization have been reported both in Finland (Saarsalmi and Mälkönen, 2001) and Sweden (Bergh et al., 2014; Sponseller et al., 2016). In contrast, irrigation experiments in the boreal region are rare. Bergh et al. (1999) found that irrigation (combined with fertilization) more than doubled the growth of Norway spruce in a fertile, southern Swedish stand. In sprinkling infiltration tree growth indicated a strong response to the added lake water; radial growth of the study trees peaked during and immediately after infiltration (Nöjd et al., 2009). Nöjd et al. (2009) accounted the observed increase in tree growth to the increased availability of N and water in the soil. They concluded, however, that pinpointing which of these two was more responsible for the growth response was difficult (Nöjd et al., 2009). Typically, these site types are both water- and nutrient-limited at the time of maximum growth, and water availability fluctuates according to season. This initial fertilizing effect of the infiltration lasted about five years after the initiation of the treatment (Figure 13).

4.5 Soil disturbance and soil carbon and long-term soil productivity

Typically, disturbance effects are manifested in the soil organic layer and top-most parts of the mineral soil. It seems that the decomposition of recently produced organic material is faster than that of the C recalcitrant pool. Mean residence time (MRT) of C in the organic layer is thus shorter than in the mineral soil (Hansson et al., 2013; Persson et al., 2017). In a coniferous stand in Sweden, Hansson et al. (2013) and Fröberg et al. (2011) reported that the MRT is 30–40 years in the organic layer and 205 years in the mineral soil, respectively. It seems that the recalcitrant soil C pool, as indicated by radiocarbon data, contributes only a minor portion of the soil CO_2 efflux,

i.e. most of the CO_2 produced during decomposition is derived from short-lived SOM (Trumbore, 2000; Giardina and Ryan, 2000). Hyvönen et al. (2007) thus concluded that soil respiration is a good indicator of metabolism, but a poor indicator of changes in the long-term soil C storage. Laboratory studies have shown that the decomposition of subsoil C with high residence time could be stimulated by addition of labile C (Rumpel and Kögel-Knabner, 2011; Karhu et al., 2016).

Disturbance such as harvesting can change the nutrient dynamics in a stand. Clear-cutting may stimulate N transformations in the soil, thus potentially increasing N losses to the atmosphere and the leaching of nitrate (NO_3-N) and base cations to waterbodies (Likens et al., 1970; Paavolainen and Smolander, 1998; Schelker et al., 2016). Increased mineralization, reduced nutrient uptake by the vegetation and changes in hydrological fluxes (groundwater table, runoff) increase the risk of N and nutrients leaching (Likens et al., 1970; Kreutzweiser et al., 2008; Schelker et al., 2016). DOC exports too usually increase after logging (Kreutzweiser et al., 2008). It is possible that in N-saturated soils (i.e. after fertilization or infiltration with surface water) microbes become C-limited and added N results in a decrease in soil respiration and soil microbial and fungal biomasses in the organic layer and thus increases in the soil C accumulation rate (Maaroufi et al., 2015). In a boreal coniferous stand in Finland, Palviainen et al. (2005a) reported a delay of two years following harvest during which the ground vegetation re-emerges and recovers and nutrient leaching losses from the soil are the greatest (Piirainen et al., 2002; Piirainen et al., 2004). A review based on North-American studies by Harmon et al. (2011) concluded that disturbance related inputs likely account for 20–50 % of all R_h losses in forests, and disturbances lead to a re-organization of ecosystem C pools, which in turn influences how R_h changes over a succession. Furthermore, within a forest stand there is a possibility of multiple periods of C sinks and sources following a disturbance (Harmon et al., 2011). Disturbance legacy too might differ between regions; boreal forests tend to change from C sources to C sinks in 5–15 years after disturbance (Harmon et al., 2011). It is likely that tree species also affect soil C but their role in engineering C into more stable form in the soil remains to be established (Vesterdal et al., 2013).

In the context of our study, these findings are interesting for a number of reasons. As described in study **I,** excavation and stump pits accumulated OM on their surface, thus potentially adding C to this layer that is vertically and horizontally very different from the soil surface. Stump harvesting creates a disturbance that vertically extends across the soil profile. Given that in upland Finnish forest soils a majority of the OM is in the first meter of the soil profile (Tamminen and Starr, 1994; Ilvesniemi et al., 2002; Callesen et al., 2003), we can assume that this pool is at least to some extent affected by stump harvesting. In a 40-year chronosequence study, Palviainen and Finér (2015) estimated that the average annual rate in which C is released from stumps and coarse roots following clear-cutting is 0.3–0.4 Mg C ha^{-1}, which over a rotation of 65 years results in 19.5–26 Mg C ha^{-1} loss, if stumps are removed. Thus, at the experimental

stand in Karkkila, it would take approximately 98–130 years for the stump and coarse root biomass (39.3 Mg C ha^{-1}) to fully decompose, had they been left in the stand (study **IV**). Therefore, the estimated 9 Mg C ha^{-1} of stump-root biomass retained in the soil after stump harvesting represents only a fraction of the temporal C pool potential. One has to acknowledge though, that the rate of decomposition of CWD is not a constant. The rate of C release (0.5–0.6 Mg C ha^{-1} yr^{-1}) from stump and coarse roots has been estimated to peak in the first 5–10 years following harvest (Grelle et al., 2012; Palviainen and Finér, 2015). In a study conducted 25 years after stump harvesting along a climatic gradient in Sweden, Strömgren et al. (2013) found a reduction in the organic layer C stock when stump harvesting was combined with logging residue harvesting, resulting in 6 Mg ha^{-1} difference with conventional, stem-only harvesting. Furthermore, considering that stumps constitute up to 80 % of the CWD on boreal whole-tree harvested clear-cuts and 28 % of the CWD on landscape level (Caruso et al., 2008; Bouget et al., 2012), stump removal inevitably results in a reduction of deadwood remaining in the stand (Eräjää et al., 2010; Anderson et al., 2015).

In contrast, sprinkling infiltration adds organic C and N and other nutrients to the soil with water, which percolates through the soil the surface layers to the vadose zone. At a different infiltration site in Southern Finland (Lindroos et al., 2001), also infiltrated with lake water in the 1990s, an estimated 10 Mg of C was added to the vadose zone in one year (Helmisaari et al., 2003). In this study (study **III**), the median N concentration (NH_4-N 0.002 mg/l, NO_3-N 0.07 mg/l and N_{tot} 0.38 mg/l) of the added lake water was relatively low, however due to the large quantities added, the annual C, N and nutrient build-up in the soil can be significant (Helmisaari et al., 2003). Paavolainen et al. (2000b) studied the N concentrations of the percolate water at the same site as Lindroos et al. (2001) and found that NH_4-N remained low throughout the infiltration. However, they observed peaks in NO_2+NO_3-N concentrations in the percolate water during breaks in infiltration (Paavolainen et al., 2000b). Thus, it seems that the actual risk that sprinkling infiltration poses to groundwater quality depends on the size of the infiltration area in relation to the whole aquifer but nevertheless needs to be monitored (Paavolainen et al., 2000b). The N concentrations of the percolate water and groundwater were measured during the infiltration in 1999–2001 at our study site too (Helmisaari et al., 2003). Throughout the treatment and after its termination NO_3-N and DON (dissolved organic nitrogen) levels in the groundwater remained below the EU-limit value of (0.5 mg/l) (Helmisaari et al., 2003). Low nitrate values are also explained by the fact that nitrification did not intensify on this site (study **III**), as discussed earlier.

Finnish upland soils are currently considered C sinks, estimated to accumulate 0.12 ± 0.06 Mg C ha^{-1} annually (Rantakari et al., 2012), however the strength of the sink varies significantly with stand age (Kolari et al., 2004; Häkkinen et al., 2011). Recent research has highlighted the need to manage C flows rather than pools in the soil as organic compounds exist in a continuum and decomposition is affected by a complex combination of environmental and biotic factors (Schmidt et al., 2011; Lehmann and

Kleber, 2015). Studying a single stock, i.e. soil C pool, at a given point in time, in a whole ecosystem is not always meaningful, as the stocks change over the whole rotation (Strömgren et al., 2013; Egnell et al., 2015). In addition, the processes and fluxes driving the stock change should be taken into consideration (Clarke et al., 2015). Heterogeneity of soil extends to its surface. A review study by Yanai (2003b) concluded that the studies they surveyed in USA and Canada had been unable to detect statistically significant changes in forest floor C or mass smaller than 15–20 %. The forest floor is very dynamic in space and time, and the appearance of disturbance changes over time and it becomes harder to identify the different disturbance classes, even with a soil corer. Re-emerging vegetation in part contributes to the changes on the soil surface, adding OM into the soils surface. Tamminen (2012) concluded that possible differences between harvest treatments may be masked by the cumulative deposition of litter. Thus over time, identifying different disturbance classes, even with a soil corer, can become more challenging and results in differences between experiments.

Finally, one has to consider the legacy of any treatment that affects forest ecosystem and particularly the soil. From a forest management perspective maintaining long-term stand productivity is crucial, particularly in the northern latitudes where rotation times can exceed 100 years. A global meta-analysis on the effects of WTH on the growth of the next tree generation by Achat et al. (2015) showed a 3–7 % growth reduction in the short to medium-term. However, as highlighted by Engell (2017) their data was focused on stands that were aged 10 years or younger, thus the result is not entirely applicable to Nordic forests with long rotations. In considering what stands could be more suitable for stump harvesting, Egnell (2016) concluded that Scots pine stands should be targeted before Norway spruce stands and that stumps should be targeted before logging residues in Norway spruce forests. The findings of Iwald et al. (2012) indicate that harvesting stumps in addition to the stem and logging residues constitutes 13–24 % of total excess cation extraction, depending on harvest intensity and tree species. Therefore, from a strict nutrient balance and acidification perspective, stump harvesting is a better alternative to logging residue harvesting (Iwald et al., 2012; Karlsson and Tamminen, 2013; Persson et al., 2017). However, the interactions of tree species and soil C and N stocks are complex (Hansson et al., 2013). Artificial groundwater recharge in the form of sprinkling infiltration adds water not only to the topsoil but also deeper into the soil. The added water percolates through the soil profile, into the realm of subsoil which is difficult to access with current soil sampling methods and yet plays a crucial role in groundwater dynamics, and which is a vital part of the critical zone.

Evidently, sprinkling infiltration caused chemical changes in the forest soil and contributed to the changes in the understory vegetation and also affected tree growth in the experimental stand. All these observations should be taken into consideration when planning new infiltration plants – areas that should preferably remain in their original state should be avoided. Finally, scientific experiments should not exist in a vacuum.

Many experiments tend to utilize carefully controlled and monitored experimental set-ups. This study in contrast was planned in cooperation with local waterworks (study **III**) and forestry companies (studies **I**,**II**,**IV**) who are responsible for the operational activities in the experimental stands, so as to avoid producing results that are not applicable in practice. Given that both the soil treatments – large scale stump harvesting and sprinkling infiltration – have been introduced relatively recently in Finland (in the last 15–20 years), one has to remark that the results of this dissertation are based on an observation period shorter than the current forest management and most probably does not cover the time in which changes occur in forest soil. Consequently, it would be ideal, however rarely possible, to have longer-term studies that cover an entire forest rotation and in addition, instead of studying the response of one variable try to address a variety of ecosystem components.

Nevertheless, the results of this study provide valuable insight into the physical and chemical changes occurring in forest soil following a disturbance, and a novel look into the biomass dynamics of stump-root systems.

5 CONCLUSIONS

Forest soils form the basis for plant-growth in forested ecosystems and have a crucial role in maintaining forest health and long-term forest stand productivity. Human-induced disturbances result in changes in C and N processes in the forest floor and the soil beneath it. Concern over how human-induced disturbances affect soil and stand C sequestration and ultimately the C budget of forested stands in the northern hemisphere has increased in the last century.

The aim of this thesis was to study the effects of physical and chemical disturbance on forest soil. Two contrasting forest soil treatments practiced in Finland – stump harvesting and sprinkling infiltration – were studied. In contrast to stump harvesting where C and nutrients are removed from the soil, sprinkling infiltration adds large quantities of C and nutrient-rich surface water onto the forest soil. Stump harvesting changes the physical structure of the forest soil, whereas sprinkling infiltration alters the chemical composition of the soil. Furthermore, in sprinkling infiltration the water percolates deep into the subsoil and causes erosion on the soil surface. Stump harvesting, in turn, creates a landscape of different surfaces that differ in topography and composition.

The results indicate that stump harvesting causes soil mixing and relocation of organic matter in the soil profile. More specifically, the findings of this study indicate that stump harvesting combined with site preparation tends to cause more extensive soil surface disturbance than site preparation alone and the mixing effect of stump harvesting persist on soil surface after a decade since harvest. In this study, we confirmed that stumps, coarse roots and fine coarse roots represent a significant portion of the belowground biomass and N in a forested stand. Furthermore, stump and coarse

roots are the largest coarse woody debris component in the managed boreal stand, thus their extraction results in direct removal of biomass and C from the stand and decreases the amount of deadwood in managed forests.

The effects of sprinkling infiltration were studied in an experimental stand that had been infiltrated with surface water in the end of the 1990s. The findings of this study indicate that the previously observed changes in soil chemistry had persisted at the experimental site; soil pH and the concentrations of base cations Ca^{2+} and Mg^{2+} had all remained high at the infiltrated plots more than a decade after the termination of the infiltration treatment. These results lead to the conclusion that sprinkling infiltration results in the long-term neutralization of the water-treated forest soil. Soil recovery can be a relatively slow process, particularly on sites where the soil cation exchange capacity has largely shifted from its original state. In a Finnish coniferous stand, acidity is added to the soil through precipitation and weathering. Tree growth peaked during and within a five-year-period after the initiation of the infiltration but had since tapered off. In contrast to tree growth, vegetation had not benefited from the added nutrients and organic matter, instead the large amounts of added water had created conditions unfavorable to certain plant species. Sprinkling infiltration had disturbed the course of natural vegetation development and favored species not typical for a sub-xeric Scots pine stand; early-successional herbs and grasses as well as deciduous shrubs, and in turn, negatively affected late successional, slow-growing mosses and lichens. Based on all the findings above, it can be concluded that sprinkling infiltration is an environment altering water and soil treatment method which, based on the findings of this study, can have short-term effects on tree growth and long-term effects on soil processes and understory vegetation and ultimately, ecosystem recovery.

The results of this study demonstrate that disturbances affect the function and structure of forest soil and these changes can persist on the surface of the soil and deeper in the mineral soil. From the perspective of this study the critical questions that remain are that to what extent – in an even longer time perspective – the studied stands are able to either compensate for the C losses following stump harvesting or persist with the C and N additions brought upon by the infiltration treatment. This study provides valuable insight into the current discourse into the sustainability of forest soil management practices and provides important information for practitioners such as foresters and land- and water-use planners.

REFERENCES

Achat, D.L., Fortin, M., Landmann, G., Ringeval, B., Augusto, L., 2015. Forest soil carbon is threatened by intensive biomass harvesting. Scientific Reports 5, 1–10. doi:10.1038/srep15991

Amundson, R., Berhe, A.A., Hopmans, J.W., Olson, C., Sztein, A.E., Sparks, D.L., 2015. Soil and human security in the 21st century. Science 348, 1261071–1261071. doi:10.1126/science.1261071

Anderson, J., Hjältén, J., Dynesius, M., 2015. Wood-inhabiting beetles in low stumps, high stumps and logs on boreal clear-cuts: implications for dead wood management. PloS ONE 10, 1–14. doi:10.1371/journal.pone.0118896

Asikainen, A., Liiri, H., Peltola, S., Karjalainen, T., Laitila, J., 2008. Forest energy potential in Europe (EU27). Working Papers of the Finnish Forest Research Institute, 69, 33.

Astrup, R., Bernier, P.Y., Genet, H., Lutz, D.A., Bright, R.M., 2018. A sensible climate solution for the boreal forest. Nature Climate Change 8, 11–12. doi:10.1038/s41558-017-0043-3

Augusto, L., Achat, D.L., Bakker, M.R., Bernier, F., Bert, D., Danjon, F., Khlifa, R., Meredieu, C., Trichet, P., 2015. Biomass and nutrients in tree root systems–sustainable harvesting of an intensively managed Pinus pinaster (Ait.) planted forest. GCB Bioenergy 7, 231–243. doi:10.1111/gcbb.12127

Berg, S., Nordfjell, T., Bergström, D., 2015. Effect of stump size and timing of stump harvesting on ground disturbance and root breakage diameter. Silva Fennica 49, 1–17. doi:10.14214/sf.1312

Bergh, J., Linder, S., Lundmark, T., Elfving, B., 1999. The effect of water and nutrient availability on the productivity of Norway spruce in northern and southern Sweden. Forest Ecology and Management 119, 51–62. doi:10.1016/S0378-1127(98)00509-X

Bergh, J., Nilsson, U., Allen, H.L., Johansson, U., Fahlvik, N., 2014. Long-term responses of Scots pine and Norway spruce stands in Sweden to repeated fertilization and thinning. Forest Ecology and Management 320, 118–128. doi:10.1016/j.foreco.2014.02.016

Bergholm, J., Olsson, B.A., Vegerfors, B., Persson, T., 2015. Nitrogen fluxes after clear-cutting. Ground vegetation uptake and stump/root immobilisation reduce N leaching after experimental liming, acidification and N fertilisation. Forest Ecology and Management 342, 64–75. doi:10.1016/j.foreco.2015.01.009

Bergstedt, J., Milberg, P., 2001. The impact of logging intensity on field-layer vegetation in Swedish boreal forests. Forest Ecology and Management 154, 105–115. doi:10.1016/S0378-1127(00)00642-3

Binkley, D., Fisher, R.F., 2012. Ecology and Management of Forest Soils. 4th edition. Wiley-Blackwell, Chichester, UK.

Bouget, C., Lassauce, A., Jonsell, M., 2012. Effects of fuelwood harvesting on biodiversity — a review focused on the situation in Europe. Canadian Journal of Forest Research 42, 1421–1432. doi:10.1139/x2012-078

Brady, N.C., Weil, R.R., 2008. The nature and properties of soils. 14th edition. Pearson-Prentice Hall, Upper Saddle River, New Jersey.

Brassard, B.W., Chen, H.Y.H., Bergeron, Y., Paré, D., 2011. Coarse root biomass allometric equations for Abies balsamea, Picea mariana, Pinus banksiana, and

Populus tremuloides in the boreal forest of Ontario, Canada. Biomass and Bioenergy 35, 4189–4196. doi:10.1016/j.biombioe.2011.06.045

Cajander, A.K., 1949. Forest types and their significance. Acta Forestalia Fennica 56, 1–71.

Callesen, I., Liski, J., Raulund-Rasmussen, K., Olsson, M.T., Tau-Strand, L., Vesterdal, L., Westman, C.J., 2003. Soil carbon stores in Nordic well-drained forest soils-relationships with climate and texture class. Global Change Biology 9, 358–370. doi:10.1046/j.1365-2486.2003.00587.x

Callesen, I., Raulund-Rasmussen, K., Westman, C.J., Tau-Strand, L., 2007. Nitrogen pools and C:N ratios in well-drained Nordic forest soils related to climate and soil texture. Boreal Environment Research 12, 681–692.

Caruso, A., Rudolphi, J., Thor, G., 2008. Lichen species diversity and substrate amounts in young planted boreal forests: A comparison between slash and stumps of Picea abies. Biological Conservation 141, 47–55. doi:10.1016/j.biocon.2007.08.021

Clarke, N., Gundersen, P., Jonsson-Belyazid, U., Kjonaas, O.J., Persson, T., Sigurdsson, B.D., Stupak, I., Vesterdal, L., 2015. Influence of different tree-harvesting intensities on forest soil carbon stocks in boreal and northern temperate forest ecosystems. Forest Ecology and Management 351, 9–19.

Deluca, T.H., Boisvenue, C., 2012. Boreal forest soil carbon: distribution, function and modelling. Forestry 85, 161–184.

Derome, Helmisaari, H.-S., Korpela, L., Lindroos, A.-J., Lumme, I., Nieminen, T.M., Nöjd, P., 2004. Sadetusimeytyksen vaikutukset Vuonteenharjulla - maaperän ja kasvillisuuden toipuminen imeytyksestä. METLA, Vantaan tutkimuskeskus, Vantaa. (In Finnish)

Derome, J., Lindroos, A.-J., Helmisaari, H.-S., 2006. Effect of sprinkling infiltration on soil acidity and fertility properties on a forested esker in Central Finland. UNESCO IHO-VI Series on Groundwater 13, 264–268.

Dirnböck, T., Grandin, U., Bernhardt-Römermann, M., Beudert, B., Canullo, R., Forsius, M., Grabner, M.T., Holmberg, M., Kleemola, S., Lundin, L., Mirtl, M., Neumann, M., Pompei, E., Salemaa, M., Starlinger, F., Staszewski, T., Uzieblo, A.K., 2014. Forest floor vegetation response to nitrogen deposition in Europe. Global Change Biology 20, 429–440. doi:10.1111/gcb.12440

Egnell, G., Jurevics, A., Peichl, M., 2015. Negative effects of stem and stump harvest and deep soil cultivation on the soil carbon and nitrogen pools are mitigated by enhanced tree growth. Forest Ecology and Management 338, 57–67.

Egnell, G., 2016. Effects of slash and stump harvesting after final felling on stand and site productivity in Scots pine and Norway spruce. Forest Ecology and Management 371, 42–49. doi:10.1016/j.foreco.2016.03.006

Egnell, G., 2017. A review of Nordic trials studying effects of biomass harvest intensity on subsequent forest production. Forest Ecology and Management 383, 27–36. doi:10.1016/j.foreco.2016.09.019

Eräjää, S., Halme, P., Kotiaho, J.S., Markkanen, A., Toivanen, T., 2010. The volume and composition of dead wood on traditional and forest fuel harvested clear-cuts. Silva Fennica 44, 203–211.

Ericsson, K., Huttunen, S., Nilsson, L.J., Svenningsson, P., 2004. Bioenergy policy and market development in Finland and Sweden. Energy Policy 32, 1707–1721. doi:10.1016/S0301-4215(03)00161-7

Finér, L., Mannerkoski, H., Piirainen, S., Starr, M., 2003. Carbon and nitrogen pools in old-growth, Norway spruce mixed forest in eastern Finland and changes associated with clear-cutting. Forest Ecology and Management 174, 51–63.

FMI, 2018. Statistics from 1961 onwards - Finnish Meteorological Institute (FMI). http://en.ilmatieteenlaitos.fi/statistics-from-1961-onwards (Accessed 1/2018)

Fröberg, M., Tipping, E., Stendahl, J., Clarke, N., Bryant, C., 2011. Mean residence time of O horizon carbon along a climatic gradient in Scandinavia estimated by 14C measurements of archived soils. Biogeochemistry 104, 227–236. doi:10.1007/s10533-010-9497-3

Gauthier, S., Bernier, P., Kuuluvainen, T., Shvidenko, A.Z., Schepaschenko, D.G., 2015. Boreal forest health and global change. Science 349, 819–822. doi:10.1126/science.aaa9092

Giardina, C.P., Ryan, M.G., 2000. Evidence that decomposition rates of organic carbon in mineral soil do not vary with temperature. Nature 404, 858–861. doi:10.1038/35009076

Grelle, A., Strömgren, M., Hyvönen, R., 2012. Carbon balance of a forest ecosystem after stump harvest. Scandinavian Journal of Forest Research 27, 762–773.

Hakkila, P., 1975. Kanto- ja juuripuun kuoriprosentti, puuaineen tiheys ja asetoniuutteitten määrä. [Bark percentage, basic density and amount of acetone extractives in stump and root wood]. Folia Forestalia 224, 14 (In Finnish)

Hakkila, P., 1989. Utilization of Residual Forest Biomass. Springer Series in Wood Science, Springer Berlin Heidelberg. doi:10.1007/978-3-642-74072-5_8

Hansson, K., Olsson, B., Olsson, M., Johansson, U., Kleja, D.B., Kleja, 2011. Differences in soil properties in adjacent stands of Scots pine, Norway spruce and silver birch in SW Sweden. Forest Ecology and Management 262, 522–530. doi:10.1016/j.foreco.2011.04.021

Hansson, K., Fröberg, M., Helmisaari, H.-S., Kleja, D.B., Olsson, B.A., Olsson, M., Persson, T., 2013. Carbon and nitrogen pools and fluxes above and below ground in spruce, pine and birch stands in southern Sweden. Forest Ecology and Management 309, 28–35. doi:10.1016/j.foreco.2013.05.029

Harmon, M.E., Bond-Lamberty, B., Tang, J., Vargas, R., 2011. Heterotrophic respiration in disturbed forests: A review with examples from North America. Journal of Geophysical Research - Biogeosciences 116, 1–17. doi:10.1029/2010JG001495

Hedwall, P.O., Brunet, J., Nordin, A., Bergh, J., 2013. Changes in the abundance of keystone forest floor species in response to changes of forest structure. Journal of Vegetation Science 24, 296–306.

Hellsten, S., Helmisaari, H.-S., Melin, Y., Skovsgaard, J.P., Kaakinen, S., Kukkola, M., Saarsalmi, A., Petersson, H., Akselsson, C., 2013. Nutrient concentrations in stumps and coarse roots of Norway spruce, Scots pine and silver birch in Sweden, Finland and Denmark. Forest Ecology and Management 290, 40–48.

Helmisaari, H.-S., Illmer, K., Hatva, T., Lindroos, A.-J., Miettinen, I., Pääkkönen, J., Reijonen, R. (Eds.), 2003. Tekopohjaveden muodostaminen: imeytystekniikka, maaperäprosessit ja veden laatu, TEMU-tutkimushankkeen loppuraportti. Metsäntutkimuslaitoksen tiedonantoja 902 [Finnish Forest Research Institute, Research Papers], Vantaa, Finland. (In Finnish)

Helmisaari, H.-S., Illmer, K., Derome, J., Hatva, T., Kitunen, V., Lindroos, A.-J., Miettinen, I., Pääkkönen, J., Reijonen, R., 2006. Artificial recharge in Finland

through basin and sprinkling infiltration: soil processes, retention time and water quality. UNESCO IHO-VI Series on Groundwater 13, 280–285.

Helmisaari, H.-S., Derome, J., Nojd, P., Kukkola, M., 2007. Fine root biomass in relation to site and stand characteristics in Norway spruce and Scots pine stands. Tree Physiology 27, 1493–1504. doi:10.1093/treephys/27.10.1493

Helmisaari, H.-S., Kaarakka, L., Olsson, B.A., 2014. Increased utilization of different tree parts for energy purposes in the Nordic countries. Scandinavian Journal of Forest Research 29, 312–322.

Hope, G.D., 2007. Changes in soil properties, tree growth, and nutrition over a period of 10 years after stump removal and scarification on moderately coarse soils in interior British Columbia. Forest Ecology and Management 242, 625–635. doi:10.1016/j.foreco.2007.01.072

Hyvönen, R., Ågren, G.I., Linder, S., Persson, T., Cotrufo, M.F., Ekblad, A., Freeman, M., Grelle, A., Janssens, I.A., Jarvis, P.G., Kellomäki, S., Lindroth, A., Loustau, D., Lundmark, T., Norby, R.J., Oren, R., Pilegaard, K., Ryan, M.G., Sigurdsson, B.D., Strömgren, M., van Oijen, M., Wallin, G., 2007. The likely impact of elevated [CO_2], nitrogen deposition, increased temperature and management on carbon sequestration in temperate and boreal forest ecosystems: a literature review. New Phytologist 173, 463–480. doi:10.1111/j.1469-8137.2007.01967.x

Häkkinen, M.N., Heikkinen, J., Mäkipää, R., 2011. Soil carbon stock increases in the organic layer of boreal middle-aged stands. Biogeosciences 8, 1279–1289.

Högberg, M.N., Blaško, R., Bach, L.H., Hasselquist, N.J., Egnell, G., Näsholm, T., Högberg, P., 2014. The return of an experimentally N-saturated boreal forest to an N-limited state: observations on the soil microbial community structure, biotic N retention capacity and gross N mineralisation. Plant and Soil 381, 45–60. doi:10.1007/s11104-014-2091-z

Högberg, P., Näsholm, T., Franklin, O., Högberg, M.N., 2017. Tamm Review: On the nature of the nitrogen limitation to plant growth in Fennoscandian boreal forests. Forest Ecology and Management 403, 161–185. doi:10.1016/j.foreco.2017.04.045

Ilvesniemi, H., Forsius, M., Finér, L., Holmberg, M., Kareinen, T., Lepistö, K., Piirainen, S., Pumpanen, J., Rankinen, K., Starr, M., Tamminen, P., Ukonmaanaho, L., Vanhala, P., 2002. Carbon and nitrogen storages and fluxes in Finnish forest ecosystems, in: Käyhkö, J., Talve, L. (Eds.), Understanding the Global System, the Finnish Perspective, 69–82.

Iwald, J., Löfgren, S., Stendahl, J., Karltun, E., 2012. Acidifying effect of removal of tree stumps and logging residues as compared to atmospheric deposition. Forest Ecology and Management 290, 49–58. doi:10.1016/j.foreco.2012.06.022

Jandl, R., Lindner, M., Vesterdal, L., Bauwens, B., Baritz, R., Hagedorn, F., Johnson, D.W., Minkkinen, K., Byrne, K.A., 2007. How strongly can forest management influence soil carbon sequestration? Geoderma 137, 253–268. doi:10.1016/j.geoderma.2006.09.003

Jurevics, A., Peichl, M., Olsson, B.A., Strömgren, M., Egnell, G., 2016. Slash and stump harvest have no general impact on soil and tree biomass C pools after 32–39 years. Forest Ecology and Management 371, 33–41. doi:10.1016/j.foreco.2016.01.008

Kaila, A., Ihalainen, A., 2014. Metsävarat [Forest Resources], in: Ylitalo, E. (Ed.), Finnish Statistical Yearbook of Forestry 2013 33–46.

Kalliokoski, T., Nygren, P., Sievänen, R., 2008. Coarse root architecture of three boreal tree species growing in mixed stands. Silva Fennica 42, 189–210.

Karhu, K., Hilasvuori, E., Fritze, H., Biasi, C., Nykänen, H., Liski, J., Vanhala, P., Heinonsalo, J., Pumpanen, J., 2016. Priming effect increases with depth in a boreal forest soil. Soil Biology and Biochemistry 99, 104–107. doi:10.1016/j.soilbio.2016.05.001

Karlsson, K., Tamminen, P., 2013. Long-term effects of stump harvesting on soil properties and tree growth in Scots pine and Norway spruce stands. Scandinavian Journal of Forest Research 28, 550–558.

Kataja-aho, S., Fritze, H., Haimi, J., 2011a. Short-term responses of soil decomposer and plant communities to stump harvesting in boreal forests. Forest Ecology and Management 262, 379–388. doi:10.1016/j.foreco.2011.04.002

Kataja-aho, S., Saari, E., Fritze, H., Haimi, J., 2011b. Effects of stump removal on soil decomposer communities in undisturbed patches of the forest floor. Scandinavian Journal of Forest Research 26, 221–231. doi:10.1080/02827581.2011.560183

Kataja-aho, S., Pennanen, T., Lensu, A., Haimi, J., 2012a. Does stump removal affect early growth and mycorrhizal infection of spruce (Picea abies) seedlings in clear-cuts? Scandinavian Journal of Forest Research 27, 746–753. doi:10.1080/02827581.2012.715185

Kataja-aho, S., Smolander, A., Fritze, H., Norrgard, S., Haimi, J., 2012b. Responses of soil carbon and nitrogen transformations to stump removal. Silva Fennica 46, 169–179.

Kaye, T.N., Blakeley-Smith, M., Thies, W.G., 2008. Long-term effects of post-harvest stump removal and N-fertilization on understory vegetation in Western USA forests. Forest Ecology and Management 256, 732–740. doi:10.1016/j.foreco.2008.05.028

Kieloaho, A.-J., Pihlatie, M., Dominguez Carrasco, M., Kanerva, S., Parshintsev, J., Riekkola, M.-J., Pumpanen, J., Heinonsalo, J., 2016. Stimulation of soil organic nitrogen pool: The effect of plant and soil organic matter degrading enzymes. Soil Biology and Biochemistry 96, 97–106. doi:10.1016/j.soilbio.2016.01.013

Kolari, P., Pumpanen, J., Rannik, Ü., Ilvesniemi, H., Hari, P., Berninger, F., 2004. Carbon balance of different aged Scots pine forests in Southern Finland. Global Change Biology 10, 1106–1119. doi:10.1111/j.1529-8817.2003.00797.x

Korhonen, J.F.J., Pihlatie, M., Pumpanen, J., Aaltonen, H., Hari, P., Levula, J., Kieloaho, A.-J., Nikinmaa, E., Vesala, T., Ilvesniemi, H., 2013. Nitrogen balance of a boreal Scots pine forest. Biogeosciences 10, 1083–1095.

Kortesmaa, T., Jokela, A., Salo-Kauppinen, R. (Eds.), 2017. Ruoka- ja luonnonvaratilastojen e-vuosikirja 2017. LUKE (Natural Resources Institute Finland), 98. (Available at :http:–stat.luke.fi–sites–default–files–luke–luobio_81_2017.pdf) (In Finnish)

Kreutzweiser, D.P., Hazlett, P.W., Gunn, J.M., 2008. Logging impacts on the biogeochemistry of boreal forest soils and nutrient export to aquatic systems: A review. Environmental Reviews 16, 157–179. doi:10.1139/A08-006

Kuuluvainen, T., Aapala, K., Ahlroth, P., Kuusinen, M., 2002. Principles of ecological restoration of boreal forested ecosystems: Finland as an example. Silva Fennica 36, 409–422.

Kuuluvainen, T., Tahvonen, O., Aakala, T., 2012. Even-aged and uneven-aged forest management in boreal Fennoscandia: a review. Ambio 41, 720–737.

Kätkö, T.S., Lipponen, M.A., Rönkä, E.K.T., 2006. Groundwater use and policy in community water supply in Finland. Hydrogeology Journal 14, 69–78. doi:10.1007/s10040-004-0351-3

Laitila, J., Asikainen, A., Anttila, P., 2008a. Energiapuuvarat, in: Kuusinen, M., Ilvesniemi, H. (Eds.), Energiapuun Ympäristövaikutukset - Tutkimusraportti [an Environmental Assessment Report on the Effects of Fuelwood Harvest in Finland] 6–12. (In Finnish)

Laitila, J., Ranta, T., Asikainen, A., 2008b. Productivity of stump harvesting for fuel. International Journal of Forest Engineering 19, 37–47. doi:10.1080/14942119.2008.10702566

Lehmann, J., Kleber, M., 2015. The contentious nature of soil organic matter. Nature 528, 60–68. doi:10.1038/nature16069

Levy-Booth, D.J., Prescott, C.E., Grayston, S.J., 2014. Microbial functional genes involved in nitrogen fixation, nitrification and denitrification in forest ecosystems. Soil Biology and Biochemistry 75, 11–25. doi:10.1016/j.soilbio.2014.03.021

Likens, G.E., Bormann, F.H., Johnson, N.M., Fisher, D.W., Pierce, R.S., 1970. Effects of Forest Cutting and Herbicide Treatment on Nutrient Budgets in the Hubbard Brook Watershed-Ecosystem. Ecological Monographs 40, 23–47.

Lindroos, A.-J., Derome, J., Paavolainen, L., Helmisaari, H.-S., 2001. The effect of lake-water infiltration on the acidity and base cation status of forest soil. Water Air and Soil Pollution 131, 153–167.

LUKE, 2017. Natural Resources Institute Finland - Statistics Database. http://stat.luke.fi/en/energy-2016_en-0 (Accessed 1/2018)

Maaroufi, N.I., Nordin, A., Hasselquist, N.J., Bach, L.H., Palmqvist, K., Gundale, M.J., 2015. Anthropogenic nitrogen deposition enhances carbon sequestration in boreal soils. Global Change Biology 21, 3169–3180. doi:10.1111/gcb.12904

Magnusson, R.I., Tietema, A., Cornelissen, J.H.C., Hefting, M.M., Kalbitz, K., 2016. Tamm Review: Sequestration of carbon from coarse woody debris in forest soils. Forest Ecology and Management 377, 1–15. doi:10.1016/j.foreco.2016.06.033

Mahli, Y., Baldocchi, D.D., Jarvis, P.G., 1999. The carbon balance of tropical, temperate and boreal forests. Plant, Cell and Environment 22, 715–740.

Manninen, O.H., Stark, S., Kytöviita, M.M., Lampinen, L., Tolvanen, A., 2009. Understorey plant and soil responses to disturbance and increased nitrogen in boreal forests. Journal of Vegetation Science 20, 311–322. doi:10.1111/j.1654-1103.2009.05529.x

Melin, Y., Petersson, H., Nordfjell, T., 2009. Decomposition of stump and root systems of Norway spruce in Sweden-A modelling approach. Forest Ecology and Management 257, 1445–1451.

Merilä, P., Mustajärvi, K., Helmisaari, H.-S., Hilli, S., Lindroos, A.-J., Nieminen, T.M., Nöjd, P., Rautio, P., Salemaa, M., Ukonmaanaho, L., 2014. Above- and below-ground N stocks in coniferous boreal forests in Finland: Implications for sustainability of more intensive biomass utilization. Forest Ecology and Management 311, 17–28. doi:10.1016/j.foreco.2013.06.029

Metcalfe, D.B., Eisele, B., Hasselquist, N.J., 2013. Effects of nitrogen fertilization on the forest floor carbon balance over the growing season in a boreal pine forest. Biogeosciences 10, 8223–8231.

Mjöfors, K., Strömgren, M., Nohrstedt, H.Ö., Gärdenäs, A.I., 2015. Impact of site-preparation on soil-surface CO_2 fluxes and litter decomposition in a clear-cut in Sweden. Silva Fennica 49, 1–20.

Mustajärvi, K., Merilä, P., Derome, J., Lindroos, A.-J., Helmisaari, H.-S., Nöjd, P., Ukonmaanaho, L., 2008. Fluxes of dissolved organic and inorganic nitrogen in

relation to stand characteristics and latitude in Scots pine and Norway spruce stands in Finland. Boreal Environmental Research 13, 3–21.

Nave, L.E., Vance, E.D., Swanston, C.W., Curtis, P.S., 2010. Harvest impacts on soil carbon storage in temperate forests. Forest Ecology and Management 259, 857–866. doi:10.1016/j.foreco.2009.12.009

Niemelä, J., 1999. Management in relation to disturbance in the boreal forest. Forest Ecology and Management 115, 127–134.

Näsholm, T., Högberg, P., Franklin, O., Metcalfe, D., Keel, S.G., Campbell, C., Hurry, V., Linder, S., Högberg, M.N., 2013. Are ectomycorrhizal fungi alleviating or aggravating nitrogen limitation of tree growth in boreal forests? New Phytologist 198, 214–221. doi:10.1111/nph.12139

Nöjd, P., Lindroos, A.-J., Smolander, A., Derome, J., Lumme, I., Helmisaari, H.-S., 2009. Artificial recharge of groundwater through sprinkling infiltration: Impacts on forest soil and the nutrient status and growth of Scots pine. Science of the Total Environment 407, 3365–3371. doi:10.1016/j.scitotenv.2009.01.062

Olesen, P.O., 1971. The water displacement method. Forest Tree Improvement 3, 3–23.

Olsson, B., Staaf, H., 1995. Influence of harvesting intensity of logging residues on ground vegetation in coniferous forests. Journal of Applied Ecology 32, 640–654.

Paavolainen, L., Smolander, A., 1998. Nitrification and denitrification in soil from a clear-cut norway spruce (Picea abies) stand. Soil Biology and Biochemistry 30, 775–781.

Paavolainen, L., 1999. Nitrogen transformations in boreal forest soils in response to extreme manipulation treatments. Finnish Forest Research Institute, Research Papers 746, Vantaa, Finland.

Paavolainen, L., Fox, M., Smolander, A., 2000a. Nitrification and denitrification in forest soil subjected to sprinkling infiltration. Soil Biology and Biochemistry 32, 669–678.

Paavolainen, L., Smolander, A., Lindroos, A.-J., Derome, J., Helmisaari, H.-S., 2000b. Nitrogen transformations and losses in forest soil subjected to sprinkling infiltration. Journal of Environmental Quality 29, 1069–1074.

Page-Dumroese, D.S., Harvey, A.E., Jurgensen, M.F., Amaranthus, M.P., 1998. Impacts of soil compaction and tree stump removal on soil properties and outplanted seedlings in northern Idaho, USA. Canadian Journal of Forest Research 78, 29–34. doi:10.4141/S97-022

Palviainen, M., Finér, L., Kurka, A.-M., Mannerkoski, H., Piirainen, S., Starr, M., 2004. Decomposition and nutrient release from logging residues after clear-cutting of mixed boreal forest. Plant and Soil 263, 53–67.

Palviainen, M., Finér, L., Mannerkoski, H., Piirainen, S., Starr, M., 2005a. Changes in the above- and below-ground biomass and nutrient pools of ground vegetation after clear-cutting of a mixed boreal forest. Plant and Soil 275, 157–167. doi:10.1007/s11104-005-1256-1

Palviainen, M., Finér, L., Mannerkoski, H., Piirainen, S., Starr, M., 2005b. Responses of ground vegetation species to clear-cutting in a boreal forest: aboveground biomass and nutrient contents during the first 7 years. Ecological Research 20, 652–660. doi:10.1007/s11284-005-0078-1

Palviainen, M., Finér, L., Laiho, R., Shorohova, E., Kapitsa, E., Vanha-Majamaa, I., 2010. Carbon and nitrogen release from decomposing Scots pine, Norway spruce and silver birch stumps. Forest Ecology and Management 259, 390–398.

Palviainen, M., Finér, L., 2015. Decomposition and nutrient release from Norway spruce coarse roots and stumps – A 40-year chronosequence study. Forest Ecology and Management 358, 1–11. doi:10.1016/j.foreco.2015.08.036

Palviainen, M., Pumpanen, J., Berninger, F., Ritala, K., Duan, B., Heinonsalo, J., Sun, H., Koster, E., Koster, K., 2017. Nitrogen balance along a northern boreal forest fire chronosequence. PloS ONE 12, 1–21. doi:10.1371/journal.pone.0174720

Pan, Y., Birdsey, R.A., Fang, J., Houghton, R., Kauppi, P.E., Kurz, W.A., Phillips, O.L., Shvidenko, A., Lewis, S.L., Canadell, J.G., Ciais, P., Jackson, R.B., Pacala, S.W., McGuire, A.D., Piao, S., Rautiainen, A., Sitch, S., Hayes, D., 2011. A large and persistent carbon sink in the world's forests. Science 333, 988–993. doi:10.1126/science.1201609

Persson, T., Lenoir, L., Vegerfors, B., 2017. Long-term effects of stump harvesting and site preparation on pools and fluxes of soil carbon and nitrogen in central Sweden. Scandinavian Journal of Forest Research 32, 222–229. doi:dx.doi.org/10.1080/02827581.2016.1218043

Persson, T., Egnell, G., 2018. Stump harvesting for bioenergy: A review of climatic and environmental impacts in northern Europe and America. Wiley Interdisciplinary Reviews: Energy and Environment 290, 1–26. doi:10.1002/wene.307

Piirainen, S., Finér, L., Mannerkoski, H., Starr, M., 2002. Effects of forest clear-cutting on the carbon and nitrogen fluxes through podzolic soil horizons. Plant and Soil 239, 301–311.

Piirainen, S., Finér, L., Mannerkoski, H., Starr, M., 2004. Effects of forest clear-cutting on the sulphur, phosphorus and base cation fluxes through podzolic soil horizons. Biogeochemistry 69, 405–424.

Piirainen, S., Finér, L., Starr, M., 2015. Changes in forest floor and mineral soil carbon and nitrogen stocks in a boreal forest after clear-cutting and mechanical site preparation. European Journal of Soil Science 66, 735–743. doi:10.1111/ejss.12264

Pirinen, P., Simola, H., Aalto, J., Kaukoranta, J.-P., Karlsson, P., Ruuhela, R., 2012. Tilastoja Suomen ilmastosta 1981–2010 [Statistics on the Finnish Climate 1981–2010]. Ilmatieteen laitos, 2012:1, 96. (In Finnish).

Prescott, C.E., Maynard, D.G., Laiho, R., 2000. Humus in northern forests: friend or foe? Forest Ecology and Management 133, 23–36.

Priha, O., Smolander, A., 1997. Microbial biomass and activity in soil and litter under Pinus sylvestris, Picea abies and Betula pendula at originally similar field afforestation sites. Biology and Fertility of Soils 24, 45–51. doi:10.1007/BF01420219

Priha, O., Smolander, A., 1999. Nitrogen transformations in soil under Pinus sylvestris, Picea abies and Betula pendula at two forest sites. Soil Biology and Biochemistry 31, 965–977. doi:10.1016/S0038-0717(99)00006-1

Pumpanen, J., Westman, C.J., Ilvesniemi, H., 2004. Soil CO_2 efflux from podzolic forest soil before and after forest clear-cutting and site preparation. Boreal Environment Research 9, 199–212.

R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria.

Rabinowitch-Jokinen, R., Vanha-Majamaa, I., 2010. Immediate effects of logging, mounding and removal of logging residues and stumps on coarse woody debris in managed boreal Norway spruce stands. Silva Fennica 44, 51–62.

Rantakari, M., Lehtonen, Linkosalo, T., Tuomi, M., Tamminen, P., Heikkinen, J., Liski, J., Mäkipää, R., Ilvesniemi, H., Sievänen, R., 2012. The Yasso07 soil carbon model – Testing against repeated soil carbon inventory. Forest Ecology and Management 286, 137–147. doi:10.1016/j.foreco.2012.08.041

Rantala, J., Saarinen, V.-M., Hallongren, H., 2010. Quality, productivity and costs of spot mounding after slash and stump removal. Scandinavian Journal of Forest Research 25, 507–514. doi:10.1080/02827581.2010.522591

Repo, A., Tuomi, M., Liski, J., 2011. Indirect carbon dioxide emissions from producing bioenergy from forest harvest residues. GCB Bioenergy 3, 107–115. doi:10.1111/j.1757-1707.2010.01065.x

Repo, A., Känkänen, R., Tuovinen, J.P., Antikainen, R., Tuomi, M., Vanhala, P., Liski, J., 2012. Forest bioenergy climate impact can be improved by allocating forest residue removal. GCB Bioenergy 4, 202–212.

Repo, A., Tuovinen, J.P., Liski, J., 2015. Can we produce carbon and climate neutral forest bioenergy? GCB Bioenergy 7, 253–262. doi:10.1111/gcbb.12134

Repola, J., 2009. Biomass equations for Scots pine and Norway spruce in Finland. Silva Fennica 43, 625–647.

Ross, D.S., Matschonat, G., Skyllberg, U., 2008. Cation exchange in forest soils: the need for a new perspective. European Journal of Soil Science 59, 1141–1159. doi:10.1111/j.1365-2389.2008.01069.x

Rudolphi, J., Strengbom, J., 2016. No support for long-term effects of commercial tree stump harvest on understory vegetation. Forest Ecology and Management 371, 84–89. doi:10.1016/j.foreco.2016.01.039

Rumpel, C., Kögel-Knabner, I., Bruhn, F., 2002. Vertical distribution, age, and chemical composition of organic, carbon in two forest soils of different pedogenesis. Organic Geochemistry 33, 1131–1142.

Rumpel, C., Kögel-Knabner, I., 2011. Deep soil organic matter-a key but poorly understood component of terrestrial C cycle. Plant and Soil 338, 143–158. doi:10.1007/s11104-010-0391-5

Saarinen, V.M., 2006. The effects of slash and stump removal on productivity and quality of forest regeneration operations - preliminary results. Biomass and Bioenergy 30, 349–356.

Saarsalmi, A., Mälkönen, E., 2001. Forest fertilization research in Finland: a literature review. Scandinavian Journal of Forest Research 16, 514–535. doi:10.1080/02827580152699358

Saarsalmi, A., Starr, M., Hokkanen, T., Ukonmaanaho, L., Kukkola, M., Nöjd, P., Sievänen, R., 2007. Predicting annual canopy litterfall production for Norway spruce (Picea abies (L.) Karst.) stands. Forest Ecology and Management 242, 578–586. doi:10.1016/j.foreco.2007.01.071

Saksa, T., 2013. Regeneration after stump harvesting in southern Finland. Forest Ecology and Management 290, 79–82.

Schelker, J., Sponseller, R., Ring, E., Högbom, L., Löfgren, S., Laudon, H., 2016. Nitrogen export from a boreal stream network following forest harvesting: seasonal nitrate removal and conservative export of organic forms. Biogeosciences 13, 1–12. doi:10.5194/bg-13-1-2016

Schimel, J.P., Bennett, J., 2004. Nitrogen mineralization: Challenges of a changing paradigm. Ecology 85, 591–602. doi:10.1890/03-8002

Schmidt, M.W.I., Torn, M.S., Abiven, S., Dittmar, T., Guggenberger, G., Janssens, I.A., Kleber, M., Kögel-Knabner, I., Lehmann, J., Manning, D.A.C., Nannipieri, P.,

Rasse, D.P., Weiner, S., Trumbore, S.E., 2011. Persistence of soil organic matter as an ecosystem property. Nature 478, 49. doi:10.1038/nature10386

Seidl, R., Thom, D., Kautz, M., Martin-Benito, D., Peltoniemi, M., Vacchiano, G., Wild, J., Ascoli, D., Petr, M., Honkaniemi, J., Lexer, M.J., Trotsiuk, V., Mairota, P., Svoboda, M., Fabrika, M., Nagel, T.A., Reyer, C.P.O., 2017. Forest disturbances under climate change. Nature Climate Change 7, 395–402. doi:10.1038/nclimate3303

Shorohova, E., Ignatyeva, O., Kapitsa, E., Kauhanen, H., Kuznetsov, A., Vanha-Majamaa, I., 2012. Stump decomposition rates after clear-felling with and without prescribed burning in southern and northern boreal forests in Finland. Forest Ecology and Management 263, 74–84. doi:10.1016/j.foreco.2011.09.006

Smolander, A., Paavolainen, L., Mälkönen, E., 2000. C and N transformations in forest soil after mounding for regeneration. Forest Ecology and Management 134, 17–28.

Smolander, A., Kitunen, V., 2011. Comparison of tree species effects on microbial C and N transformations and dissolved organic matter properties in the organic layer of boreal forests. Applied Soil Ecology 49, 224–233. doi:10.1016/j.apsoil.2011.05.002

Smolander, A., Kanerva, S., Adamczyk, B., Kitunen, V., 2012. Nitrogen transformations in boreal forest soils-does composition of plant secondary compounds give any explanations? Plant and Soil 350, 1–26. doi:10.1007/s11104-011-0895-7

Sponseller, R.A., Gundale, M.J., Futter, M., Ring, E., Nordin, A., Näsholm, T., Laudon, H., 2016. Nitrogen dynamics in managed boreal forests: Recent advances and future research directions. Ambio 45, 175–187. doi:10.1007/s13280-015-0755-4

Strandström, M., 2006. Effect of stump lifting on sprouting at the regeneration site., in:. Presented at the NSFP seminar - Forest regeneration and bioenergy, Vantaa, Finland.

Strengbom, J., Nordin, A., Näsholm, T., Ericson, L., 2001. Slow recovery of boreal forest ecosystem following decreased nitrogen input. Functional Ecology 15, 451–457. doi:10.1046/j.0269-8463.2001.00538.x

Strömgren, M., Mjöfors, K., 2012. Soil-CO_2 flux after patch scarification, harrowing and stump harvest in a hemi-boreal forest. Scandinavian Journal of Forest Research 27, 754–761.

Strömgren, M., Egnell, G., Olsson, B.A., 2013. Carbon stocks in four forest stands in Sweden 25 years after harvesting of slash and stumps. Forest Ecology and Management 290, 59–66. doi:10.1016/j.foreco.2012.06.052

Strömgren, M., Mjöfors, K., Olsson, B.A., 2017. Soil-surface CO2 flux during the first 2 years after stump harvesting and site preparation in 14 Swedish forests. Scandinavian Journal of Forest Research 32, 213–221. doi:10.1080/02827581.2016.1221993

Sucre, E.B., Fox, T.R., 2009. Decomposing stumps influence carbon and nitrogen pools and fine-root distribution in soils. Forest Ecology and Management 258, 2242–2248. doi:10.1016/j.foreco.2009.05.012

Tamminen, P., Starr, M., 1994. Bulk density of forested mineral soils. Silva Fennica 28, 53–60.

Tamminen, P., Saarsalmi, A., Smolander, A., Kukkola, M., Helmisaari, H.-S., 2012. Effects of logging residue harvest in thinnings on amounts of soil carbon and nutrients in Scots pine and Norway spruce stands. Forest Ecology and Management 263, 31–38. doi:10.1016/j.foreco.2011.09.015

Tarvainen, O., Hekkala, A.-M., Kubin, E., Tamminen, P., Murto, T., Tolvanen, A., 2015. Soil disturbance and early vegetation response to varying intensity of energy wood harvest. Forest Ecology and Management 348, 153–163.

Thiffault, E., Hannam, K.D., Paré, D., Titus, B.D., Hazlett, P.W., Maynard, D.G., Brais, S., 2011. Effects of forest biomass harvesting on soil productivity in boreal and temperate forests — A review. Environmental Reviews 19, 278–309. doi:10.1139/a11-009

Tonteri, T., Salemaa, M., Rautio, P., Hallikainen, V., Korpela, L., Merilä, P., 2016. Forest management regulates temporal change in the cover of boreal plant species. Forest Ecology and Management 381, 115–124. doi:10.1016/j.foreco.2016.09.015

Trumbore, S., 2000. Age of soil organic matter and soil respiration: radiocarbon constraints on belowground C dynamics. Ecological Applications 10, 399–411. doi:10.2307/2641102

Vesterdal, L., Clarke, N., Sigurdsson, B.D., Gundersen, P., 2013. Do tree species influence soil carbon stocks in temperate and boreal forests? Forest Ecology Management 309, 4–18. doi:10.1016/j.foreco.2013.01.017

Walmsley, J.D., Godbold, D.L., 2010. Stump harvesting for bioenergy - a review of the environmental impacts. Forestry 83, 17–38.

Wild, B., Schnecker, J., Knoltsch, A., Takriti, M., Mooshammer, M., Gentsch, N., Mikutta, R., Alves, R.J., Gittel, A., Lashchinskiy, N., Richter, A., 2015. Microbial nitrogen dynamics in organic and mineral soil horizons along a latitudinal transect in western Siberia. Global Biogeochemical Cycles 29, 567–582. doi:10.1002/2015GB005084

Yanai, R.D., Currie, W.S., Goodale, C.L., 2003a. Soil carbon dynamics after forest harvest: An ecosystem paradigm reconsidered. Ecosystems 6, 197–212. doi:10.1007/s10021-002-0206-5

Yanai, R.D., Stehman, S.V., Arthur, M.A., Prescott, C.E., Friedland, A.J., Siccama, T.G., Binkley, D., 2003b. Detecting change in forest floor carbon. Soil Science Society of America Journal 67, 1583–1593.

Zabowski, D., Chambreau, D., Rotramel, N., Thies, W.G., 2008. Long-term effects of stump removal to control root rot on forest soil bulk density, soil carbon and nitrogen content. Forest Ecology and Management 255, 720–727. doi:10.1016/j.foreco.2007.09.046

Zanchi, G., Pena, N., Bird, N., 2012. Is woody bioenergy carbon neutral? A comparative assessment of emissions from consumption of woody bioenergy and fossil fuel. GCB Bioenergy 4, 761–772. doi:10.1111/j.1757-1707.2011.01149.x